5-22-2014

Sustainability Informed Management of End-of-Life Photovoltaics: Assessing Environmental and Economic Tradeoffs of Collection and Recycling

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SUSTAINABILITY INFORMED MANAGEMENT OF END-OF-LIFE PHOTOVOLTAICS: ASSESSING ENVIRONMENTAL AND ECONOMIC TRADEOFFS OF COLLECTION AND RECYCLING

by

MICHELE GOE

A DISSERTATION

Submitted in partial fulfillment of the requirements
for the degree of Doctor of Philosophy
in
Sustainability

Department of Sustainability

Golisano Institute for Sustainability
Rochester Institute of Technology
May 22, 2014

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By

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Submitted by Michele Goe in partial fulfillment of the requirements for the degree of Doctor of Philosophy in Sustainability and accepted on behalf of the Rochester Institute of Technology by the dissertation committee.

We, the undersigned members of the Faculty of the Rochester Institute of Technology, certify that we have advised and/or supervised the candidate on the work described in this dissertation. We further certify that we have reviewed the dissertation manuscript and approve it in partial fulfillment of the requirements of the degree of Doctor of Philosophy in Sustainability.

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SUSTAINABILITY PROGRAM
ROCHESTER INSTITUTE OF TECHNOLOGY
May 22, 2014

ABSTRACT

Golisano Institute of Sustainability
Rochester Institute of Technology

Degree Doctor of Philosophy **Program** Sustainability

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Title: Sustainability Informed Management of End-of-Life Photovoltaics: Assessing Environmental and Economic Tradeoffs of Collection and Recycling

Renewable energy technologies have emerged to address the negative environmental impacts of increasing use of fossil fuels. Solar photovoltaics (PV) are an attractive renewable energy technology because they avoid significant carbon emissions during use common to non-renewables, have a long useful lifetime estimated at 20 – 30 years, and they take advantage of a stable and plentiful energy resource – the sun. However, it has been suggested that material availability is a potential constraint for broad deployment of PV. For example, solar PV's core technology depends on several primary materials i.e. indum and tellurium which were recently determined to be of high importance for the development of a clean energy economy and at near-critical supply risk. In order to evaluate the risks to supply, the environment, and the economy a broader definition of criticality that goes beyond physical scarcity to include sustainability metrics e.g. embodied energy, political instability, economic value was developed. Using this methodology several policies are suggested that depart from traditional command-and-control approaches. One criticality mitigating strategy, material recycling, is at odds with current PV research where there is a strong emphasis on efficiency gains. Recycling is a strategy with potential that has yet to be fully recognized due to the current lack of collection infrastructure and uncertain set of processing technologies. This work explores under what conditions the energy payback time (EPBT) of PV modules containing recycled materials demonstrate equivalent energy savings to improvements in efficiency. These EPBT improvements from recycling motivate further methodological work on the economically optimal PV recycling infrastructure. This methodology includes a case study that demonstrates model sensitivity in addition to revealing important tradeoffs for recycling policy and economics.

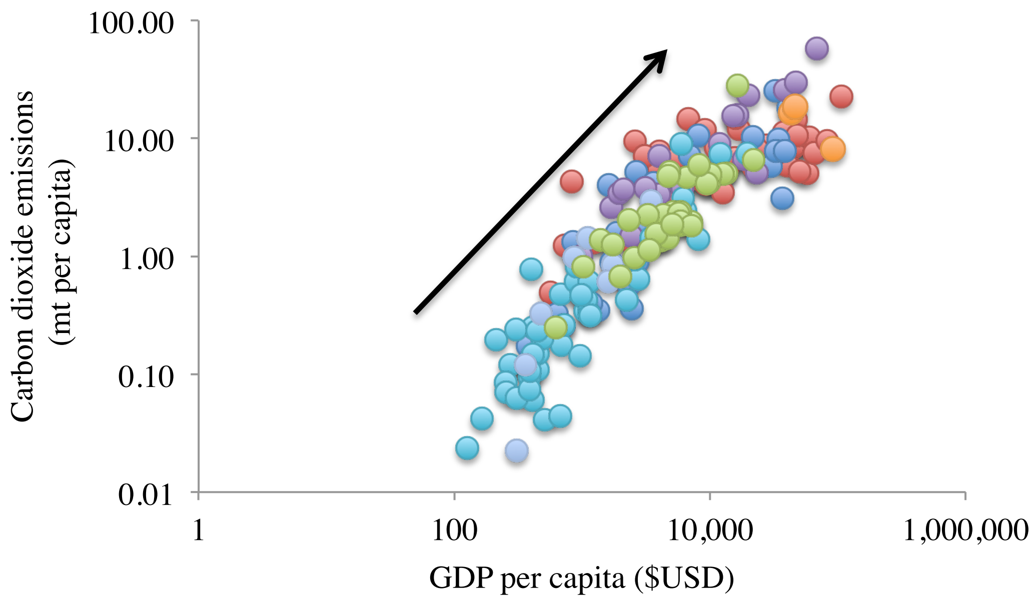
TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

1. INTRODUCTION

The United Nations Brutland Commission defines sustainable development as "meeting the needs of the present without compromising the needs of the future[1]." This definition implies that in order to meet essential needs of humans e.g. food, housing, healthcare, water, we must seek a quality of life that is "within the planet's ecological means." However, on the path to sustainable development, there are formidable challenges. These challenges include the link between global economic growth and carbon dioxide emissions, energy use, and material consumption. That is, the more carbon dioxide we emit (Figure 1.1a) the more energy (Figure 1.1b) and materials (Figure 1.2) we consume, the higher our GDP per capita. Increases in per capita material consumption and energy use occur despite technology efficiency gains. In addition, the Earth's population is expected to double in 40 years to 9 billion in 2050[2]. Each of these factors multiplies the challenge to sustainability by creating complex and potentially catastrophic impacts such as climate change (Figure 1.3) and resource scarcity. In order to mitigate these potentially catastrophic environmental impacts, research has focused on developing technology that makes use of non-polluting and non-exhaustible or renewable resources. Renewable resources i.e. wind, solar, biomass, geothermal and hydro promise a lower environmental burden and greater availability than traditional fossil fuels. Of these, solar has the greatest availability with a theoretical energy capacity one thousand times our current energy needs (Figure 1.4).



(a)

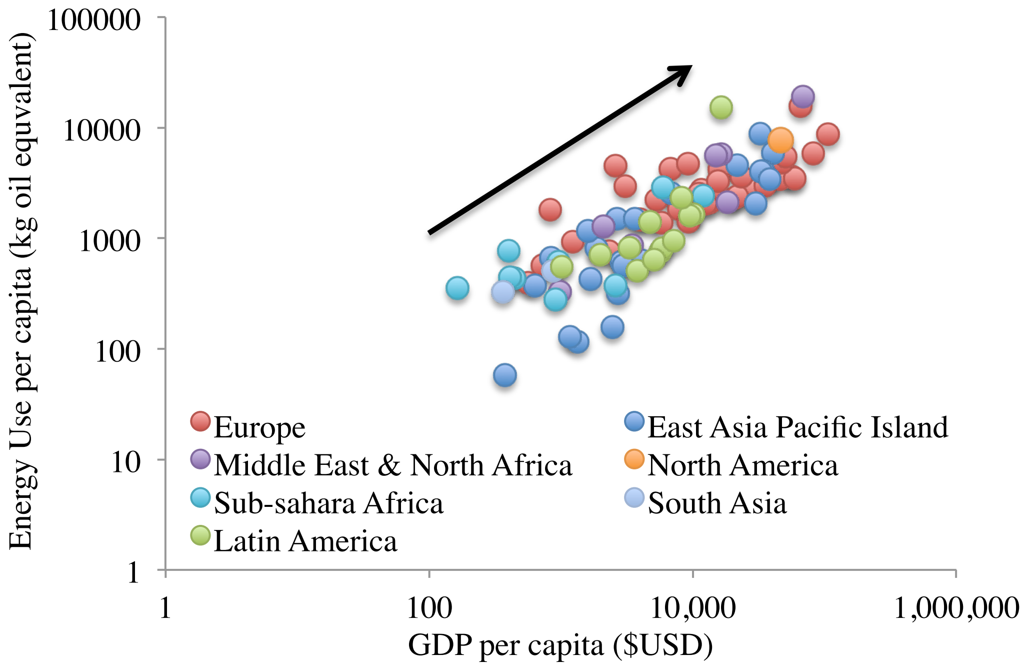


Figure 1.1 (a) Carbon dioxide emissions (b) Energy use per capita as function of GDP per capita. Data Source: [3]

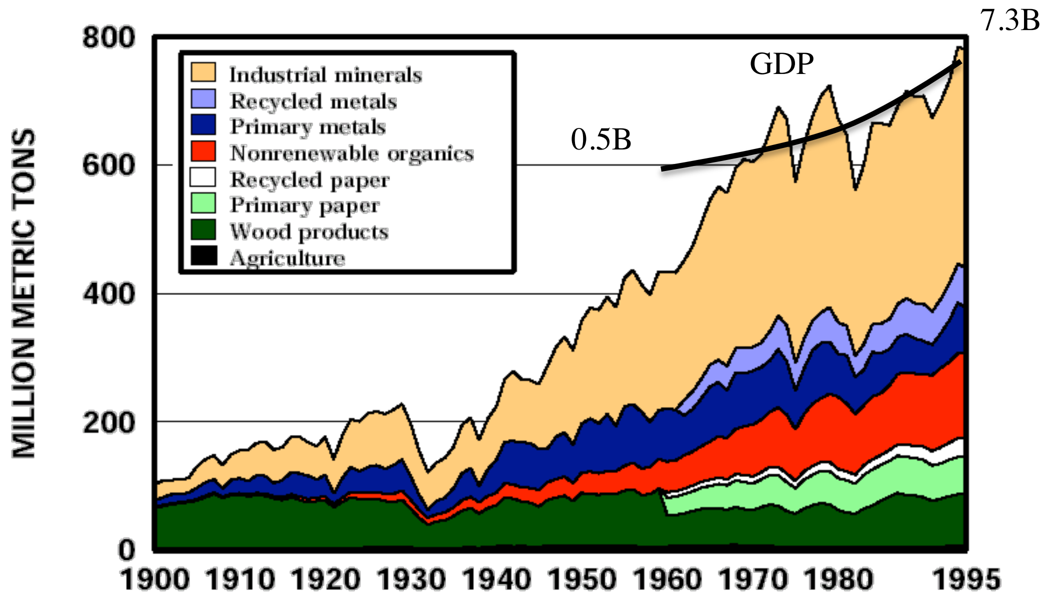


Figure 1.2 Historical US Materials Consumption and GDP [4]

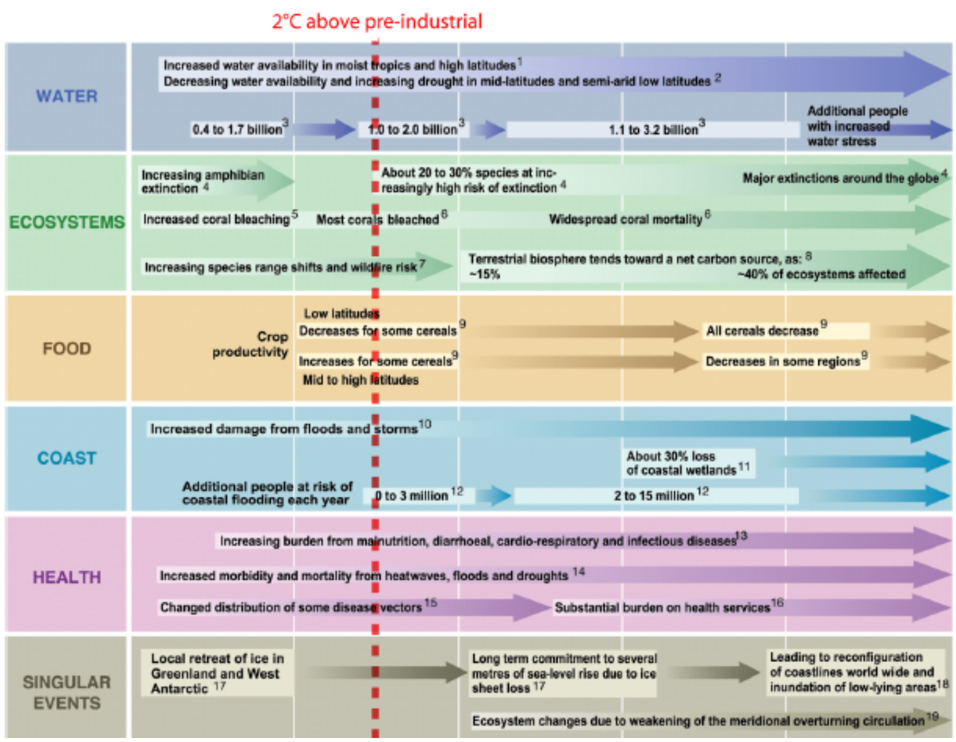


Figure 1.3 Climate Change Impacts beyond 2°C Warming[5]

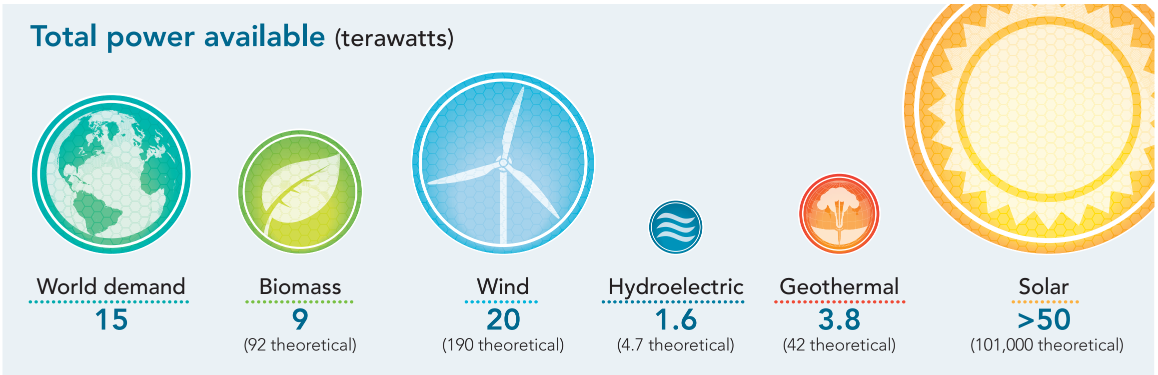


Figure 1.4 Renewable Energy Power Availability by Source[6]

Solar is a renewable energy whose technologies have the promise of environmental, economic and social benefits over current fossil fuels. Solar technology's environmental benefits include fewer carbon emissions and a faster lifecycle energy payback than fossil fuels e.g. coal and oil and some renewables. For example, the technology lifetime required to reap the primary energy investment of material and production, or energy payback time, of solar technologies ranges form 1 – 9 years. For comparison, the energy payback of coal fire power plants and wind power is 2.5 – 5 and 18 – 34 years, respectively[7,8]. In addition, unlike fossil fuels, solar technologies emit no carbon during the use phase. It is of little surprise therefore that the total lifecycle emissions of solar are one to two orders of magnitude below that of fossil fuels on per GWh basis. The potential economic benefits of solar i.e. generation potential that outpaces energy demand and rapidly decreasing costs, are also impressive. For example, the available capacity of solar energy from the sun is theoretically over 6,000 times greater than our current energy need[6]. This capacity is more than what is estimated to be available for all other renewables i.e. wind, geothermal, biomass combined. The cost per watt has decreased by 80% over two decades (1985 – 2005) for solar PV technologies[9]. As of January 2014, solar will be less than 15 years from being cost competitive with coal [10,11]. From a social perspective, solar energy enables growth of green economy, access to electrification for rural applications, and increase health outcomes. Recent research suggests the greatest impact to carbon emissions would be obtained by replacing coal fired power plants with PV plants. Coal fire power plants also have documented hazardous emissions and negative health impacts that are not associated with solar.

Literature suggests that the economic impacts from employment in the green economy are positive [12](UNEP, 2007). This includes rural electrification projects, which have enabled access to education, safe cooking practices, and overall increase standard of living.

The promise of solar has emerged due to intensive work to develop many configurations of diverse compositions. For example solar photovoltaics (PV), consists of more than ten technologies divided into four major groups – mature silicon, thin-films, organic, and multi-junction – that collectively utilize dozens of materials. The solar PV module utilizes these materials in its multiple layers for various purposes e.g. adhesion (ethyl-vinyl acetate or EVA), structure (glass, steel, aluminum (Al)), anti-reflection - Tin (Sn), and conduction - titanium (Ti). Active material layers, the core of PV technology, target specific wavelengths of sunlight in order to capture energy and produce electricity. Active layer material compositions of commercial modules include: crystalline-silicon (c-Si), amorphous-silicon (a-Si), cadmium-telluride (CdTe), and copper-indium-gallium-diselenide (CIGS). Many other emerging compositions in development such as multi-junction combine one or more commercial compositions in multiple active layers. There are various product applications for this technology such as ground-mounted, building integrated, roof-mounted, and consumer products. Overall adoption has increased 100% per year for the last decade with PV technologies that utilize silicon (Si) i.e. multi-crystalline and mono-crystalline (c-Si) dominating market share as shown in Figure 1.5. Silicon for solar PV is obtained from waste electronic production however silicon is also increasingly produced from primary silica via the Czochralski process. Thin-film PV retains the second largest total capacity with a combined 25% share. Thin-film photovoltaics such as CdTe, CIGS, and amorphous silicon (a-Si) are commonly frameless with a glass substrate. The active materials such as cadmium (Cd), induium (In), gallium (Ga), tellurium (Te), selenium (Se) are daughter metals obtained from zinc-lead (Zn-Pb) or copper (Cu) ores as shown in Figure 1.6. The remainder market share, which is less than 1%, is made of organic PV (OPV). Other emerging PV compositions include multi-junction, which combines multiple thin-film compositions in several active layers have greater theoretical efficiency however are expected to require greater manufacturing energy.

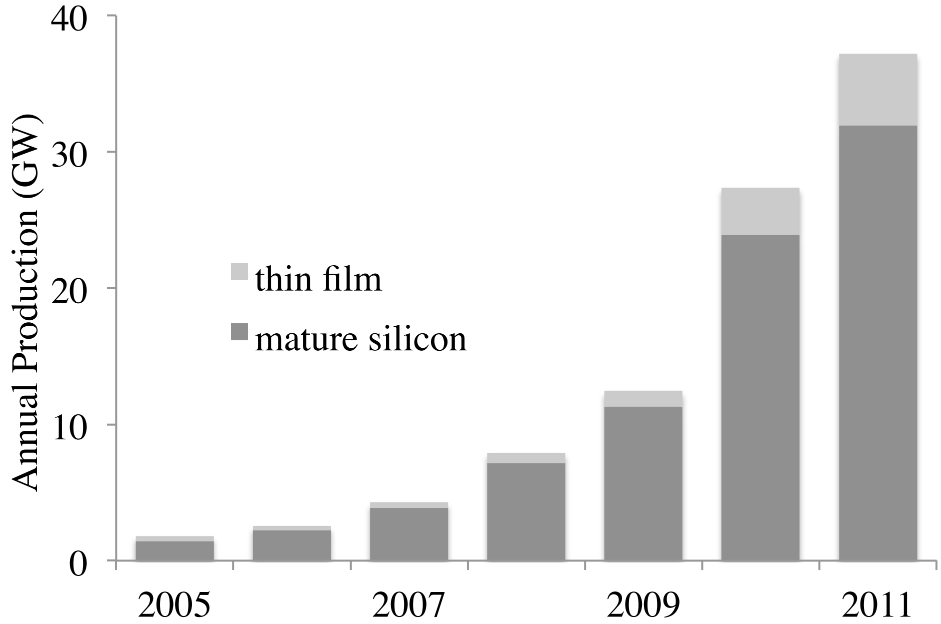


Figure 1.5 Annual and Cumulative Solar PV Production [13]

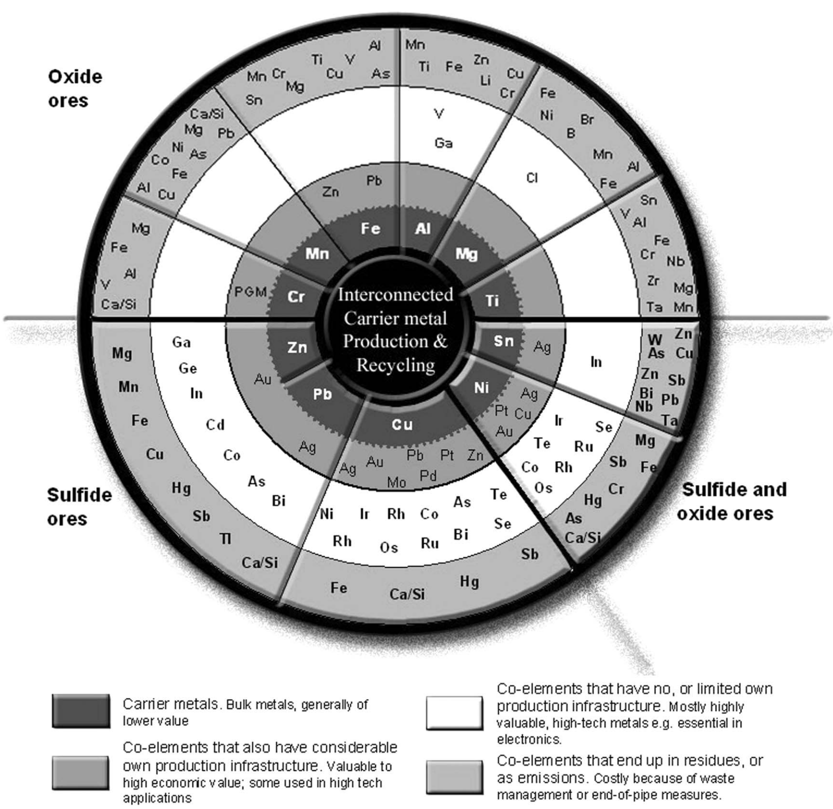


Figure 1.6 Primary and Daughter Metals[14]

Despite the environmental benefits, exponential growth in capacity and the rapid decline of prices, solar PV is not without some controversy. Several papers have expressed concern over land use burden, supply risks due to the use of energy intensive primary materials, high manufacturing energy requirements, and build up of unrecovered end-of-life waste in landfills [15-18]. This dissertation seeks to address these concerns by answering the primary research question: should PV be recycled? We answer this question from three perspectives: (1) supply risks and criticality (Chapter 2), (2) cumulative energy demand of recycling (Chapter 3), and (3) end-of-life recovery infrastructure (Chapter 4). The organizing questions that summarize the approach of each perspective are outlined in Table 1.1. Chapter 2 characterizes the material supply risks using multi-metric analysis and suggests policies to address material scarcity. One strategy proposed in Chapter 2 is recycling. Chapter 3 investigates the primary energy intensity of solar PV materials at various recycling rates and efficiencies. This work assesses the scenario of exhaustive recovery for all PV materials. Exhaustive recovery requires a recovery infrastructure. Chapter 4 outlines a methodology for economically optimal siting of a recovery infrastructure. A case study area of New York was utilized to demonstrate the methodology and perform waste policy analysis.

Table 1.1 Inquiry perspectives and organizing questions: Should PV be recycled?

Ch.	Perspective	Organizing Questions
2	Supply Risk / Criticality	(1) What metrics are useful for policy-makers in assessing and regulating criticality issues? (2) What policies would address criticality? (3) To what extent will recycling address criticality of PV materials?
3	Cumulative Energy Demand	(4) What is the impact of recycled content on the energy payback time for PV? (5) What is the material priority for recycling? (6) How does recycling compare with efficiency as a strategy for increasing energy savings?
4	Recovery Infrastructure	(7) How do we model recovery for future PV materials uncertain spatial dispersion? (8) How do we evaluate the influence of spatial and non-spatial criteria on system configuration? (9) How sensitive is our method to economic, technical, and environmental assumptions?

2. IDENTIFYING CRITICAL MATERIALS FOR PHOTOVOLTAICS IN THE US: A MULTI-METRIC APPROACH

2.1 Introduction

The United States is a dominant consumer of primary energy and materials in the world. However, the growth of emerging economies such as China and India and their increasing consumption of energy and materials have begun to draw attention towards materials availability and criticality concerns. Further deepening these concerns is the recognition of the United States' import reliance on primary energy fuels and some primary materials; of particular relevance are rare earth metals with applications in emerging electrical and energy technologies. These metals are increasingly mined in adversarial or socio-politically unstable nations [19]. One emerging technology that may be essential to US energy security and climate change mitigation is solar photovoltaics (PV). With respect to life cycle carbon emissions and land use, PV technologies have less environmental impact than traditional energy technologies i.e. coal power plants[20,21]. This implies that broad PV deployment would significantly reduce global greenhouse gas emissions and its associated climate impacts. However, it has been suggested that material availability is a potential constraint for broad deployment of PV[22-25]. For example, current silicon-based and thin-film solar PV's core technology depends on several primary materials i.e. In and Te which were recently determined to be of high importance for the development of a clean energy economy and at near-critical or critical supply risk by the US Department of Energy (DOE) [26]. Recent PV research also assesses the broader impacts of material choice[27-32].

Concerns over material availability, especially for emerging technologies, are not new and over the last several decades have sparked debates as well as national policies aimed at securing critical materials[33]. These policies continue to be implemented despite the lack of a broader definition of criticality. For example, the most recent Department of Defense (DoD) Strategic and Critical Materials report per the Strategic and Critical Materials Stockpiling Act [34]uses material consumption, production, and projected future demand to determine the severity of material criticality. Similarly, in previous literature [22,35-38], the material availability is determined primarily by physical scarcity, however, systems level considerations such as the production share of politically instable nations, toxicity, embodied energy, or the value to the economy are not considered. The use of a broader definition of criticality would likely increase the scope to include energy intensive materials such as aluminum and silicon that are not physically scarce but have broad economic and environmental implications.

2.1.1 Aims of this study

Earlier literature claimed material criticality concerns at the policy level were waning by pointing to increased foreign mineral reliance and decreased domestic mining [39-41]. Similar circumstances have motivated recent interest in identifying critical materials. Several nations including those in the European Union (EU) have recently identified materials that are common to photovoltaics e.g. In, Ga, and Ge as critical in terms of supply risk and economic importance [42-46]. However these studies lack sensitivity of results to data uncertainty and organization; they also rely on relative rather than normative determinations of criticality which lack context for (future) supply risks. For example, the Centre for Policy Related Statistics' aggregation of product groups masks supply chain dependencies. The Morley et al. study contains no clear environmental metric and aggregates similar metrics e.g. depletion time, reserve base to determine a single criticality "score" which ignores the interdependence of data. Other criticality studies have proposed methodology to ascertain the supply risk from a corporate, national, and global perspective [47,48]. Furthermore, none of the studies mentioned above address uncertainty as to the impact of a limited supply of base metals e.g. Cu, Al, Zn on the criticality of their by-product metals e.g. In, Ga, Te. Lastly, these studies are limited in the breath of criticality metrics especially with regards to economic and environmental risks which would provide policymakers with a more systemic perspective.

Several questions arise from the aforementioned literature gaps: What metrics are useful for policy-makers in assessing and regulating criticality issues? What policies would address metal criticality while at the same time continue to encourage solar PV adoption? Addressing criticality in policy is challenging due to the highly interconnected geopolitical relationships of supply chains, infrastructure lock-in, and the increasing material demand that must be balanced with low carbon supply. This work aims to quantify and compare a uniquely broad set of criticality metrics for silicon-based and thin-film i.e. cadmium telluride (CdTe), copper indium gallium diselenide (CIGS), amorphous silicon (a-Si) PV technologies that focus on a more comprehensive or life cycle systems approach which is unique in its inclusion of environmental metrics. This analysis highlights comparisons between metrics and combinations of metrics. In addition, we suggest how to depart from traditional command-and-control policies utilizing the aforementioned metrics to mitigate criticality in the short and long term.

2.1.2 Criticality definition and materials considered

Material criticality, as defined here, is a relative concept in that it compares, in this case, solar PV materials against each other to determine which materials have the greatest risks of disruption to supply and greater impacts on the economy or the environment. In order to evaluate the criticality of solar PV materials from the perspective of the US we characterize three areas of criticality: supply risk (Section 2.3.1), economic risk, (Section 2.3.2) and environmental risk (Section 2.3.3). This is a semi-dynamic study in that we include select data for materials over a 20-year period (1992 - 2012) commenting on their trends in the context of the decision making for policy. The solar PV materials considered in this study and their previously identified criticality issues are summarized in Table 2.1.

Table 2.1 Potential Critical Solar PV Metals Considered for this Study

Material	Previously Identified Criticality Issues	Source
Aluminum (Al)	Economic importance	[42,49]
	Defense/Military importance	
Arsenic (As)	Toxicity	[48]
	High Import Reliance	
Cadmium (Cd)	Toxicity	
Copper (Cu)	Defense/Military importance	[49]
Iron (Fe)	Global demand	[45]
Gallium (Ga)	Low Substitutability	[42,46,50]
	Recycling constraints	[26,45]
	Producer trade restrictions	

	Import reliance	
	Importance to “clean energy”	
	Carbon footprint of mining and production	
Germanium (Ge)	Economic importance and supply risk	[42] [50]
	Substitutability	[45]
	Carbon footprint of mining and production	
Gold (Au)	Carbon footprint of mining and production	
Indium (In)	High demand from emerging technologies	[42]
	Technical difficulty of recycling and substitution	[46,50]
	Import reliance	[26]
	Secondary production constrained	
	Importance to “clean energy”	
	Geological scarcity	
Molybdenum (Mo)	Economic importance	[42]
	Limited number of mining corporations	
	Substitutability	
Platinum (Pt)	Regional concentration of mining	[50]
	Recycling restriction	
	Rapid demand growth	
Selenium (Se)	Net import reliance	[42]
Silicon (Si)	Recycling Constrained	[45]
	Global demand	
Silver (Ag)	Toxicity	[45]
	Political instability of producers	
	Climate change vulnerability of producers	
Tellurium (Te)	Economic importance	[42]
	Recycling constraints	[26,50]
	Importance to “clean energy”	[45]
	Geological scarcity	
Zinc (Zn)	Economic importance	[42] [49]
	Defense/Military importance	[45]

Political instability of producers

2.2 Methodology

In order to evaluate the risks to supply, the environment, and the economy we quantified criticality components and their associated indicators as shown in Table 2.2. A key challenge in assessing criticality is to synthesize and appropriately weigh indicators of various scales and units. Previous studies have aggregated and weighed multiple indicators based on national priority or arbitrarily [26,51]. For a clear comparison, this work uses percentages or normalization to characterize the various criticality indicators.

Table 2.2 Criticality related risk, risk components, indicators, and data sources

Criticality related risk	Components	Indicators	Data sources
Supply	Institutional inefficiency	Net import reliance Hirfindahl-Hirshmann index of primary material and ore producers	[52,53]
	Physical scarcity	Recycling rate Ratio of production to reserves	[48,54, 55]
Environmental	Human toxicity	CERCLA points	[56,57]
	Energy intensity	Primary embodied energy Energy savings	
Economic	Material specific	Primary material price	[53,58, 59]
	Economy-wide	Domestic consumption Economic value by sector	

2.2.1 Calculation of supply risk indicators

Here supply risk metrics refer to two components of scarcity identified by Alonso et al. [60]: physical resource constraint and institutional inefficiency. The latter refers to the resource quality and the effort required to obtain it. We evaluate institutional inefficiency of a material using net import reliance and Hirfindahl-Hirshmann Index.

Net import reliance is defined as the ratio between net imports to apparent consumption, see Table 2.3 for values. Net import is defined by the US Geological Survey (USGS) as the difference between imports, exports, and stock changes. Apparent consumption is defined as the summation of production, imports, and stock changes minus exports. For some non-PV materials apparent consumption has been shown to significantly underestimate total consumption due to the imports and exports of products that contain large amounts of a material [61].

Table 2.3 Historical U.S. net import reliance of PV materials

Material	2005	2009	2010	2011	2012e
Ag	0.61	0.58	0.65	0.64	0.57
Bauxite		1.00	1.00	1.00	1.00
Al	0.45	0.10	0.14	0.20	0.03
As	1.00	1.00	1.00	1.00	1.00
Au	0.04	E	0.40	E	E
Cd	E	E	E	E	E
Cu	0.42	0.21	0.32	0.34	0.35
Fe	0.15	0.11	0.06	0.07	0.11
Ga	0.99	0.99	0.99	0.99	0.99
Ge	NA	0.90	0.90	0.90	0.90
In	1.00	1.00	1.00	1.00	1.00
Mo	<.01	<.01	<.01	<.01	<.01
Pt	0.93	0.95	0.91	0.89	0.91
Se	0.53[a]	E	E	E	E
Si	0.47	<0.50	<0.50	<0.50	<0.50
Sn	0.78	0.74	0.74	0.73	0.75
Te	W	0.77[b]	0.80[b]	W	W
Zn	0.56	0.77	0.73	0.74	0.72

Note: E indicates net exporter, W indicates withheld to avoid company proprietary data, e indicates estimated data for that year

[a]2005 value withheld 2006 value used

[b]Calculated value assuming US production of 13 Mg based on Graedel et al. [47] and no adjustments to government and industry stock changes because data are withheld
Source: USGS 2012, 2013[53,62]

We follow previous studies [42,63,64] that make use of the Hirfindahl-Hirshmann Index (HHI_{WGI}) to characterize the relative supply risk related to socio-political stability of material and mine producers. The assumption here is that the greater the socio-political stability of producers the smaller the risk of supply disruption for a given material. This index indicates the concentration of mineral production (S) obtained from countries with low World Governance Indicators (WGI) of Political Stability Absence of Violence (PSAV) scores. The WGI-PSAV utilizes survey and expert data from 30 sources to develop an index score for 213 countries. The score rates political stability on a scale from poor (-2.5) to good (+2.5). For example, in 2010, the US, ranked in the 56% percentile with a score of 0.31 ± 0.23 while China ranked in the 24% percentile with a score of -0.76 ± 0.23. We scaled and inverted the WGI score so that a score of zero indicates good stability and poor stability is a score of 10. A single score metric for such a complex characterization is challenging and some studies have cited issues with data completeness and uncertainty[65,66]. However, this metric provides a widely accepted first pass indicator for national stability.

$$\boldsymbol{HHI_{WGI} = \sum_{C} (S)^2 WGI_C}$$

Equation 2.1

The supply risk indicators related to physical scarcity are the ratio of global production to global reserves and recycling rate. The USGS defines a reserve base as resources that have a “reasonable potential for becoming economically available within planning horizons beyond those that assume proven technology and current economics.” Similarly, reserves are defined as the part of the reserve base that could be “economically extracted or produced at the time of determination.” Reserve base and reserve data is important to quantify in relation to future demand (apparent consumption) in order to extrapolate potential depletion time scales.
Where primary reserve data for PV materials is unavailable due to abundance e.g. Si or scarcity e.g. Ga it has been estimated from a combination of base metal reserves, by-

products, ore grade, and refining efficiency assumptions as shown in Table 2.4 and Table 2.5.

Table 2.4 World production and reserves data in metric tons of PV materials

Material	Reserves (metric tons)	Production (metric tons)
Ag	5.4E+05	2.33E+04
Al	7.0E+09[c]	4.44E+07
As	8.8E+05	4.58E+04
Au	5.2E+04	2.66E+03
Cd	5.0E+05	2.22E+04
Cu	6.8E+08	1.61E+07
Fe	8.0E+10	1.52E+09
Ga	5.0E+04[d]	2.92E+02
Ge	4.5E+05	1.18E+02
In	1.0E+03[e]	6.62E+02
Mo	1.1E+07	2.64E+05
Pt	6.6E+07	1.95E+02
Se	9.8E+04	1.98E+03
Si	3.3E+07[f]	7.37E+06
Sn	4.9E+06	2.44E+05
Te	2.4E+04	2.00E+02[b]
Zn	2.2E+09	1.28E+07

Source: USGS, 2012 [62]
[a]2010 data
[b]derived from Houari et al.[22] assuming supply equal to demand and US consumes Te which is used primarily as an alloy to steel, at the same rate it consumes steel i.e. 7%
[c]bauxite reserves assuming 17% refining efficiency to aluminum based on Norgate et al.[67]
[d]USGS states world resources of gallum in bauxite ore exceeds, 1 billion kg, of these we assume 5% is recoverable based on the USGS statement: “small percentage of this metal in bauxite… is economically recoverable”
[e]2008 data
[f]assuming 5 decades 3% growth per year in demand can be met by reserves, where production estimates demand based on USGS statement “World and domestic resources for making silicon metal and alloys are abundant and, in most producing countries, adequate to supply world requirements for many decades”

Table 2.5 Base metals and PV material by-products

Base metal	By-product
Bauxite	Al
Cu	Ag, As, Au, Mo, Se, Te
Fe	Fe
Ni	Pt
Silica	Si
Sn	Sn
Zn - Pb	Ag, As, Au, Ga, Ge, In, Cd

Source: USGS, 2012 [62]

2.2.2 Calculation of economic risk indicators

There are three indicators for economic risk that we quantify here: primary price, the domestic consumption, and value to the economy. The market price of a primary material is one indicator of the cost of mining, extracting, refining, and transporting materials. We assume the greater the price, the greater the impact on the economy. This is because the price signals both the product value and the cost to acquire an alternative. Here USGS average prices were used [53]. The USGS compiles average annual prices and domestic consumption for all materials as shown in Table 2.7. In order to evaluate the consumption value of each material we calculate the product of primary price and consumption. The largest market sectors by product volume for each metal were obtained from USGS and Graedel et al.[68]. The Annual Industry Accounts released by the US Bureau of Economic Analysis details the value to the economy of each industrial sector as shown in Table 2.6[69]. In order to assess the value of a material to the US economy we allocated it to an industrial sector based on its applications, as shown in Table 2.8. For most materials e.g. In product applications spanned multiple industrial sectors and therefore were allocated to multiple GDP categories.

Table 2.6 Gross domestic product (GDP) value by sector

GDP Category	Value (USD, 2012)
Manufacturing	1.7.E+12
Construction	5.3.E+11
Retail trade	9.1.E+11
Transportation and warehousing	4.5.E+11
Agriculture, forestry, fishing, and hunting	1.8.E+11
Information communication technology producing industries	6.5.E+11

Table 2.7 Primary price and domestic consumption of PV materials

Material	Primary price (2012 USD per metric ton)	Consumption (metric tons)
Ag	998,602	5.9E+03
Al	2,013	4.5E+06
As	1,698	6.7E+03
Au	53,433,576	1.5E+02
Cd	1,980	4.8E+02[a]
Cu	7,948	1.8E+03
Fe	648	1.0E+08
Ga	556,000	3.5E+01
Ge	1,380,000	4.0E+01
In	566,667	9.0E+01
Mo	27,614	1.8E+04
Pt	50,798,180	1.6E+02
Se	127,868	4.8E+02
Si	2,866	6.4E+05
Sn	20,695	4.2E+04
Te	155,000	1.3E+03
Zn	1,937	9.4E+05

a2012 data withheld 2010 data used

Table 2.8 GDP Market Sector Allocation for Products and Material Combinations

GDP Market Sector	Material - Product or Industry
Manufacturing	In – Pb free solders
	Fe – industrial machines
	Ge – catalysts
	Zn – galvanization, alloys
	Te – chemicals, alloys
	Se – alloys
	Mo – chemicals, steel, stainless
	Cd – pigments
	Ag – industrial
Construction	Fe – construction
	Al – buildings
	Se – glasses
Retail trade	Ag – photography, jewelry
	Cd – batteries
	Zn – alloys
	Se – glasses
Transportation and warehousing	Cd – batteries
	Fe – transportation
	Se – glasses
Agriculture	Se – agriculture
Information communication technology producing industries	As – photovoltaics
	Ge – fiber optics
	In – LCD (TV), monitors,
	Ga – IC, optoelectronics
	Ge – fiber optics, infrared optics
	Te – electronics
	Se – glasses

2.2.3 Calculation of environmental risk indicators

The environmental risk category quantifies a variety of impacts to human health and the natural environment resulting from the energy use, consumption, and toxicity of materials throughout their life-cycle. Primary material embodied energy (EPE) was obtained from SimaPro 7.3.3 using the Ecoinvent database v2.0 according to [57]. The energy savings is the difference between primary and secondary energy, as shown in equation 2.2. However, for several materials (i.e. Se, Cd, Zn, Te, In, Ga, Mo, As) analyzed in this paper, secondary data is not available in the Ecoinvent database. Similar to previous studies [70] regression was performed from existing secondary data. Regression coefficients *a* and *B* were found to be -0.9762 and 16.361, respectively, with high correlation to existing data ($R^2 = 1$).

$$\boldsymbol{Energy\ Savings = a(EPE) + B}$$

Equation 2.2

The risk to human health is quantified by the Environmental Protection Agency's (EPA) Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) 2011 data according to [56]. CERCLA evaluated and ranked the toxicity of 859 compounds based on three equally weighted criteria: frequency of occurrence at national priorities list (NPL) or Superfund sites, toxicity, and the potential for human exposure. Toxicity criteria score ranges from 0 to 600 and was evaluated by the compound's ignitability or reactability, aquatic toxicity, chronic toxicity, carcinogenicity, and radio-nucleotides. The total score ranges from 0 to 1800. Graedel et al. [48] have made use of ReciPE endpoints to evaluate the environmental implications of material decisions while many other criticality studies have left this aspect out completely [26,44,71,72]. Similar to our approach, Graedel et al. represents potential human and aquatic toxicity from mineral extraction and refinement stages. However, the Graedel et al. is limited because it does not include historical frequency of use or concentration in the environment. Other common environmental health and safety metrics include: permissible exposure limit (PEL), recommended exposure limit (REL), EPA carcinogen classification, reference dose (RfD), threshold limit value (TLDV). Of these, all but TLDV, which is specific to material exposure for a worker, is considered in the CERCLA toxicity score.

2.3 Criticality policy and indicators

2.3.1 Things governments can do to address criticality

There are a myriad of things that governments can to address criticality issues. We argue here that a comprehensive approach to criticality requires a critical look at the implications of using a single indicator or a particular set of narrowly focused indicators. For the PV materials studied, we find that the use of multiple economic, environmental, and supply indicators reveals potential opportunities and tradeoffs of policy actions. To demonstrate this finding, we have proposed seven policy categories, discussed throughout section 2.3, each of which highlight a previously employed policy mechanism using a single supply, economic, or environmental indicator in Table 2.9. These policies are aligned vertically along a sliding scale from direct or command-and-control to moderate strategies. Each category then has three potential policy mechanisms aligned from potentially most to least expensive.

The most direct policies of imposing critical materials import and export taxes have important consequences. On the one hand, import taxes could increase domestic production and encourage the development of substitutes for critical materials. Alternatively, import taxes could also increase the domestic price high enough to slow the US transition to renewable energy, a consequence that has broad human health and climate change impacts. Additionally, taxing imports from adversarial nations could isolate and further marginalize these societies whose economic cooperation may be one strategy to increase their political stability. Similarly, waste export tax, could increase recycling rate but also increase toxicity issues in current landfills if these materials are not properly recovered. For example, the 2002 US import tariffs on steel, has been suspected of costing more domestic steel jobs than it was protecting and jeopardizing foreign trade relations[73].

There are several direct government approaches to mitigating criticality such as those involving a framework of economic activity and financing. The former approach includes stockpiling, increasing trade with adversarial nations and increasing the defense budget to ensure the security of critical materials. For example, the presence of US aircraft carriers in the Persian Gulf from1990 thru 2010 in order to protect oil supply routes to the US[74]. Another example is China's stockpiling and restricted exports of rare earth metals[75]. Although these policies may increase the security of supply in the short run they can be prohibitively expensive. Financing is another aggressive government approach to criticality which includes mandating that a portion of energy resources, technology, or raw materials purchases be sourced from secondary materials, socio-politically stable nations, or domestically. These policies have the objective of increasing recycling rates, decreasing supply risk, decreasing net import reliance, and influencing the ratio of future consumption to reserves. However, in implementing these policies as with securing all critical material resources by expanding our national security budget, it may be prohibitively expensive despite its positive long-term supply impacts. For example, many US states have mandates on the purchase of recycled paper.

Whereas imposing a tax serves as a deterrent for certain activities, offering subsides or grants uses economic incentives to encourage activities with the same goal of reducing material criticality. A subsidy of domestic mining of critical materials would incentivize domestic production or otherwise potentially lead to an increase in prices and primary energy when reserves are more expensive to develop domestically than abroad. However, if the US has domestic expertise in mining for a particular material then, the primary material cost and embodied energy can be reduced. For example, US domestic mining subsidies for uranium are credited with enabling nuclear power generation in the US[76]. Another subsidy utilized by governments is for research and development (R&D), for example, to enable recovery and product manufacturing from secondary materials. The goal of this policy is to increase the recycling rate and availability of critical materials. Potential advantages include reduced human health impacts when materials formerly disposed in landfills are collected for end-of-life recovery. In addition, R&D investment has the potential of increasing economic activity from the creation of new markets that make use of secondary materials.

Another policy aimed at reducing supply risks without directly influencing markets is information. For example, disseminating information on a streamlined patent process for technologies that utilize substitutes. Like R&D investment this policy has the potential to increase economic activity in renewable energy sectors and other new markets. Another form of information policy is developing an online secondary materials exchange or published reports on critical material availability. There are many advantages to encouraging knowledge accumulation of the domestic supply and demand flows; the downside is the risk of exposing domestic supply vulnerabilities to potential terrorists. Education on the other hand, focuses on domestic expertise building by providing training to material recovery facilities (MRFs) in secondary recovery techniques, to local state governments on ways to mitigate climate change without the use of critical resources, and to MRFs on increasing resource recovery efficiency of critical materials. Education policies have the advantage of potentially increasing recycling rate and reducing toxicity risks. However, an unintended consequence to discouraging the use of critical materials may be the divestment from current PV technology.

Table 2.9 Things that governments can do to mitigate criticality issues of PV materials

Category	Indicator	Materials to target	Potential mechanism (example)	(least expensive)
Taxes	Import reliance Primary price	In, Ga, Pt Au, Pt, Ag, Ga	Tax imports from adversarial nations	Tax waste exports containing critical materials
Regulations	Recycling rate Reserves to Production	Si, As, Se, Te Ga, Ge, Au	Restrict export products containing critical materials	Restrict landfilling of electronics and other products containing critical materials
Framework Economic Activity	Economic value Political instability	Se, Cd, Fe, Cu Ge, Pt, Si	Increase defense budget for securing materials abroad	3-month stockpile of critical materials
Government Financing	Import reliance Political instability	In, Ga, Pt Ge, Pt, Si	Government agencies must purchase from politically stable nations	Mandate percent government technology purchases use secondary materials
Subsidies & Grants	Ratio reserves to production Primary price	Ga, Ge, Au Ga, In, Pt	Provide domestic mining subsidies	Provide R&D grant electronics recovery technologies
Education	Primary energy Recycling rate	Au, Pt, Ag, Ga Si, As, Se, Te	Teach MSW managers strategies to increase material recovery	Provide training to support recovery processing techniques
Information	CERCLA Domestic consumption	As, Cd, Zn Fe, Al, Pt, Au	Develop national secondary materials exchange online	Streamlined patent process for renewables utilizing abundant substitutes

Indirect

2.3.2 Comparison of single category metrics

Table 2.10 shows the aggregated ranking results for each of 10 indicators from 1 to 17 order of greatest to least combined economic, supply, and environmental risk. Therefo the higher the rank number, the smaller the relative criticality risks. Individual metrics described in detail in sections 3.2.1, 3.2.2 and 3.2.3. The ranking assumes equal weighting for each of the indicators, which skews the results toward supply risk considerations as they make up 40% of the total. Our analysis determined the PV materials in order of most to least critical for these metrics are: Ge, Pt, As, In, Sn, Ag, Se, Si, Te, Cd, Zn Au, Ga, Cu, Mo, Al, and Fe

Table 2.10 Criticality priority of PV materials

Rank	Mat.	HHI	Net import reliance	Recycling rate	Ratio of production to reserves	CERCLA score	Relative primary embodied energy (Fe=1)	Energy savings	Primary price ($1000/mt)	Domestic Consumption ($M)	Econ. Sector Value (portion of GDP)
1	Ge	3.28	0.90	0.3	4	255	4	0.6	1,380	55	0.16
2	Pt	3.05	0.89	0.65	85	-	10,515	0.95	50,798	8,331	0.17
3	As	2.28	1.00	<0.01	19	1,665	25	0.95	2	11	0.04
4	In	0.76	E	<0.01	2	288	2	0.63	128	61	0.29
5	Sn	0.71	0.64	0.32	20	488	166	0.97	999	5,892	0.17
6	Ag	1.90	0.73	0.22	23	608	9	0.9	21	875	0.16
7	Se	2.25	1.00	0.38	49	778	103	0.97	567	51	0.16
8	Si	0.62	0.99	0.18	53	-	121	0.97	556	19	0.04
9	Te	1.56	0.80	<0.01	120	196	6	0.87	155	31	0.16
10	Cd	1.00	0.74	0.27	23	1,319	1	0.4	2	1,824	0.17
11	Zn	3.00	<0.40	<0.01	168	919	41	0.96	3	1,834	0.16
12	Au	1.25	0.20	0.36	20	-	6	0.78	2	9,099	0.15
13	Ga	1.41	<0.01	0.33	43	112	2	0.67	28	497	0.11
14	Cu	1.56	0.07	0.41	42	805	1	0.96	1	65,448	0.21
15	Mo	0.59	E	0.18	42	442	12,511	0.98	53,434	8,015	0.1
16	Al	0.98	E	0.14	158	685	2	0.6	2	1	0.25
17	Fe	0.67	0.34	0.3	53	-	1	0.71	8	14	0.19

Our criticality designation is based on an ordinal ranking and is limited in its ability to measure how far apart two materials are. For this reason, we also plot risk indicators along an axis (Fig. 2.3-2.6) to gain a perspective about relative risk compared to normalized metrics e.g. %GDP. Unfortunately, there is no clear benchmark or line which we can draw that determines whether a material is critical or not from the US perspective. Criticality determinations ultimately depend on stakeholder priority, available information, and future demands. For example, if the US prioritizes short term access and availability of materials for national security as indicated in recent literature [49], then Se, Te and In may be determined to be non-critical materials (due to either a low import reliance or a high production to reserve ratio). Here we attempt to align PV materials relative to one another rather than to make absolute judgments on criticality.

2.3.3 Supply risk indicator results

According to the political instability indicator all of the PV materials and base metals studied have high supply risk when compared against various distribution scenarios. Scenarios were developed to serve as baselines that determine the severity of political instability and concentration of producers for each material. In the 'best' case scenario, all producers have an equal share of production and a political stability equal to that of the U.S. This means that each producer's socio-political stability (WGI score) was set equal to that of the U.S and the total production is equally divided among producers. In the 'worse' case scenario, one instable producer dominates production. Therefore each producer's socio-political stability score was set to that of China and the shares of production are distributed unequally. Historical data indicates that the trends of unequal distribution and concentration of production have become more pronounced for all PV materials over the last decade except Au, Se, Pt, and In. These trends indicate a shift in production towards single country dominance. For example, in 2012, 11 of 17 materials studied had one producer, China, which held 30 – 60% share of primary production as compared to the rest of the world (ROW), as shown in Fig. 2.1. We also found that, in general, as the concentration of production increased, the political instability indicator increased. The presence of one or several extremely unstable non-dominant producers e.g. Somalia has little impact on the overall political instability of the supply chain where single country dominance is most severe. This implies that for all materials, policies that enable equal distribution among producers is more effective at decreasing the supply risk (as measured by HHI) than any other ‘single country’ approaches e.g. encouraging more production from individual producers that are very stable or improving conditions of very unstable non-dominant producers.

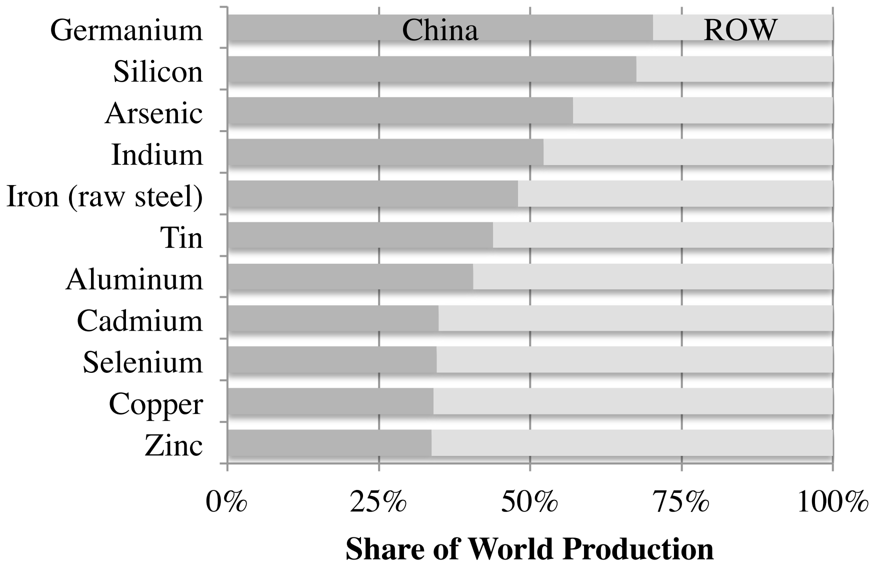


Figure 2.1 Share of world primary production held by a single producer, China, as compared to the rest of the world (ROW) of various PV materials

Similar to HHI, net import reliance (NIR) is an indicator of the quality and effort required to obtain physical resources. All PV materials except Au, Cd, Mo, Fe, and Se, are dependent on imports to meet greater than 25% of apparent consumption. Similar to global production, the bulk of domestic imports are obtained from one or two producers as shown in Fig 2.2 From 2005 – 2012, NIR of a majority of PV materials have remained constant as shown in Table 2.3; Al and Si have decreased reliance, while Zn and Ge have increased reliance. Improving NIR requires an increase in domestic production and a decrease in imports. Therefore, this indicator encourages policies that increase domestic control of key material resources e.g. protective tariffs, mining industry subsidies. In general, one would expect that greater domestic control increase security. However, in the case of unforeseen domestic supply disruptions e.g. weather, extreme adherence to this strategy would decrease security both domestically and globally. Furthermore, because NIR narrow focus on domestic supply chain quality, it provides a false sense of material security. That is, due to the complexity and interdependence of the global materials markets, a supply disruption e.g. terrorist attack can be catastrophic to the entire material supply chain despite any particular domestic reliance.

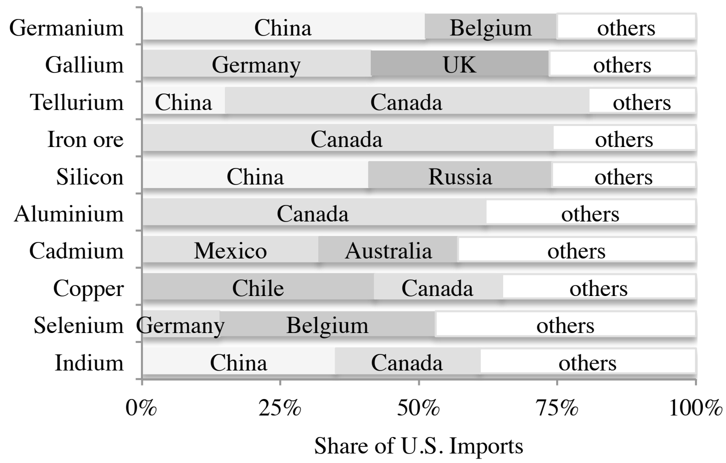


Figure 2.2 Share of US imports held by the top one or two producers

HHI and NIR are indicators for institutional inefficiency; however, more straightforward scarcity metrics i.e. ratio of reserves to production and recycling rate measure actual physical quantities of resources available. Where primary reserve data for PV materials is unavailable due to resource abundance e.g. Si or scarcity e.g. Ga it has been estimated from a combination of base metal reserves, ore grade, and refining efficiency assumptions, as discussed in Appendix A.1.3. When compared to global production, reserves are 2 to 120 times greater for most PV materials (with the exception base metals Al, Zn and Fe). If we assume no recycling, constant production, and no changes to stock or reserve estimates, reserves could be depleted in a few generations for half of the PV materials studied. Under these assumptions, In, Ag and Ge have the greatest risk of depletion in the next 20 years. Since demand is increasing, reserves are likely to be depleted even sooner. Although, increasing price, technological efficiency, the discovery of new reserves, and increasing recycling rates are all also delaying the steady march towards resource depletion. In order to further quantify supply risks due to future consumption Angerer et al. [55] posited that by 2030 consumption levels of Ga, In, and Pt would exceed current production by 2 to 6 times. Historically, future consumption of PV materials has exceeded past production by 1.8 to 2.5 times for 20 and 30-year outlooks since 1980. An exception to this steady trend is Ga whose recent consumption is nearly 11 times the production of 30 years prior. This rapid increase in demand was due the expansion of electronics e.g. computers that required Ga for integrated circuits and optoelectronics e.g. light emitting diodes (LEDs) and solar cells. The indicators of physical scarcity for primary resources discussed above can elucidate broad implications for disruptions in supply. For example, supply disruptions for Ga could have slowed development and innovation of information and communications technology (ICT) that now account for 650 billion USD or 4% of US GDP in 2012. However, the growth of new industries such as ICT is often unpredictable. Therein lays the problem of relying on physical scarcity indicators to evaluate supply risk: future uncertainty makes any assertions about depletion unreliable. Despite this uncertainty, the use of these indicators has driven aggressive policies such as stockpiling and investing in the development of new reserves that seek to avoid short-term supply disruptions due to depletion. Alternatively, physical scarcity indicators can be utilized to drive less aggressive policies such as increasing new and old scrap recycling which seek to delay depletion until more abundant substitutes are developed.

Recycling rates of base metals i.e. Zn, Fe, Cu and precious metals i.e. Au, Ag, Pt are between 27 – 41% and 18 – 65%, respectively; the highest of all PV materials. In general, as observed by Graedel et al. [77], recycling rates are more closely related to material applications (i.e. use volume and ease of recovery) than physical scarcity; materials embedded in small amounts in complex electronics e.g. Si, As have lower recycling rates than those used in large volume products with less material mixing e.g Cd in batteries and Sn in cans. Exceptions to this trend occur with In, Ga, and Ge, which is likely due to high primary (and therefore secondary) price. Use phase barriers to recycling also include the level of dissipative use, reuse in markets without a recovery infrastructure, and product lifetime. Recycling rates reflect not only the use phase but also end-of-life barriers to secondary production. For example, all materials studied have a nearly zero recycling rate with respect to its use in PV applications, despite the rapid deployment in PV applications and attention to metal criticality concerns over the last few decades. Lack of PV recycling has been attributed to low economic incentives[78], inadequate recovery technology or infrastructure[79-81], and the lack of policy incentives [82]. Shortages in base or by-product metals could impact energy security; PV recycling initiatives may be able to delay these impacts until more abundant substitutes are developed. Recognizing this opportunity, the EU has included solar PV modules in its corporate take back mandate for electronics [83] and the National Renewable Energy Laboratory (NREL) has funded research in alternative PV compositions that avoid physically scarce materials[84].

2.3.4 Economic risk indicator results

Precious metals, Au, Pt, and Ag, have the highest primary price of the PV materials studied followed by In, Ga, and Ge. As previously discussed, these metals also have greater physical and institutional scarcity issues as compared to other PV materials. We observe, in general, high volume materials e.g. Fe, Al, and Zn are typically lower priced; and low volume materials e.g. In, Ga, Ge, and Au are higher priced. Exceptions include materials with low price and low volume i.e. Si, Cd, and As. In addition, high volume materials also have higher consumption value in dollars. It is therefore expected that those materials with higher consumption will contribute more to the US economy as measured by share of sector GDP. However, this is not necessarily the case. We observe that due to the methodology chosen the more economic sectors that utilize a material, the greater a material's economic importance indicator. For example, although Ag, Cd, and Se have much lower volumes than Al they are utilized in more economic sectors and therefore are determined to have a greater share of sector GDP. Despite the limitations of this method, high volume materials such as Fe, and Zn were also determined to be among the most important materials to the economy when using share of sector GDP as an indicator. From a policy perspective, the indicators that identify materials of greatest importance to the economy can drive strategies to shield markets from the economic ramifications of lack of material availability. These strategies may include government financing of vulnerable materials markets, enforcing price controls, or offering subsidies for sectors transitioning to abundant substitutes or recycling waste materials.

2.3.5 Environmental risk indicator results

The environmental indicators measure two aspects of risk: energy intensity and toxicity, discussed below. Arsenic has been historically used as a chemical weapon, insecticide, and a medicine before being identified as a carcinogen. Cadmium dust inhalation and zinc ingestion are also toxic, having been known to cause poisoning and absorption disruption of essential minerals. Not surprisingly, materials such as As, Cd, and Zn have the greatest toxicity issues, while Pt, Si, and Fe each have the least toxicity. Most (i.e. 10 of 14) of the PV materials included in this study have CERCLA scores in the top 51 percentile of the 859 materials list. Of these, 7 materials are in the top 25 percentile. What separates the materials with more acute environmental impacts are their exposure score, a portion of the calculated CERCLA metrics. Over 6 years (from 2005 to 2011) there has only been minor movement of CERCLA scores between 1 – 5% for PV materials; Si, Ge, and Ga are the notable exceptions. Although Si maintains a ranking of 699 of 859 materials, its CERCLA score has doubled in this period of time due to its increased concentration in the environment. Both Ga and Ge have moved from up in ranking due to increases in exposure, toxicity and frequency in the environment. In and Te also saw minor CERCLA score increases due to concentration in the environment over this 6-year period. These trends show that PV active materials are becoming more prevalent at Superfund sites. Increases in domestic PV production, adoption, and end-of-life landfilling will likely increase this trend. Policy can mitigate these environmental impacts by encouraging secondary production so that materials avoid landfilling, investing in more efficient pollution control technology for production facilities, and increasing the penalty for environmental dumping of critical materials.

Primary embodied energy and energy savings (from the use of secondary materials) measure the energy intensity of environmental risk. When PV materials are compared to more common materials such as Fe, they all exhibit greater energy intensity; this is especially the case for Pt, Au, Ag, and Ga, which are several orders of magnitude greater than Fe. Zn, Cu, Cd, and Se have the lowest primary energy intensity of within 10 – 90% of Fe. In terms of energy savings, Pt, Au, Ag, and Ga yield the greatest benefits, while Zn, Ge, Cd, and Mo yield the least energy savings from the use of secondary materials. This is expected since in general, the greater the primary energy the greater the energy savings from the use of secondary materials. Energy savings are due to avoidance of high energy processing e.g. extraction from mines, electrolysis, refining that is not needed for most secondary processes which involve physical separation and re-melting. Unexpectedly we observed an empirical relationship between CERCLA score and primary energy intensity. As the primary energy intensity of a material increases the CERCLA score decreases. When coupled with material price information, this relationship may explain why less energy intensive and less expensive materials are found with greater frequency and concentration at Superfund sites despite their high toxicity e.g. As, Zn, Cu and Se: their recovery is not efficient from an energy or economic standpoint. These relationships can also motivate policy that uses economic mechanisms to drive secondary production by increasing landfill tip fees, taxing waste exports, or mandating a portion of government technology purchases meet a recycled content standard.

2.3.5 Multi-metric results

Traditional strategies aimed at addressing criticality concerns have relied on single metric command and control policies e.g. stockpiling, direct government take over of mineral producers, protective tariffs. These command-and-control policies have been criticized for their narrow focus on physical scarcity and their economic inefficiency[85-87]. Our intent is to show the use of multiple metrics with a lifecycle perspective can lead to systems level approach to criticality policy that addresses not only physical scarcity but also institutional inefficiency, environmental impacts, and economic risks.

Figure 2.3 – 2.5 shows simultaneously the economic, supply, and environmental risk of PV materials using four different sets of indicators. The x-axis for Fig. 2.3-2.4 are the same; therefore, we can observe vertical shifts and diameter size changes across figures to understand the impact of different economic, supply and environmental indicators on the overall criticality order. For Fig. 2.3-2.4, increasing economic and environmental risk is towards the top right corner of the figure. Data point diameter size changes from small to large indicate increasing supply risk. In Fig. 2.3, As, Cd, and Zn are the most toxic while precious metals Au, Pt are the least toxic but most expensive; Fe and Si are the least expensive and least toxic, in addition, Ge, In, Si and As have the highest producer socio-political instability. For this set of metrics there is a tradeoff between price and CERCLA score for PV materials. As previously observed, this relationship may explain why less energy intensive and less expensive materials e.g. As, Zn, Cu, Se are found with greater frequency and concentration at Superfund sites despite their high toxicity: their recovery is not efficient from an energy or economic standpoint. To a lesser extent there is also a relationship between price and producer political instability, in general, the higher the price the greater the producer political instability. As stated above, since price is also inversely related to domestic consumption, materials with higher political instability also have lower domestic consumption. These relationships suggest that policies aimed at reducing a single metric may impact multiple attributes of energy security. Taking a multi-metric approach to criticality policy demands, for this set of metrics, any policy aimed at, for example, reducing producer political instability is coupled with (or at least consider) sensitivity to environmental impacts to human health, and economic impacts. One such example is the promotion of sustainable mining in politically instable nations where the actions to reduce negative environmental impacts e.g. groundwater contamination of mining operations are coupled with

technology efficiency improvements, stable-living wages, and community stakeholder involvement.

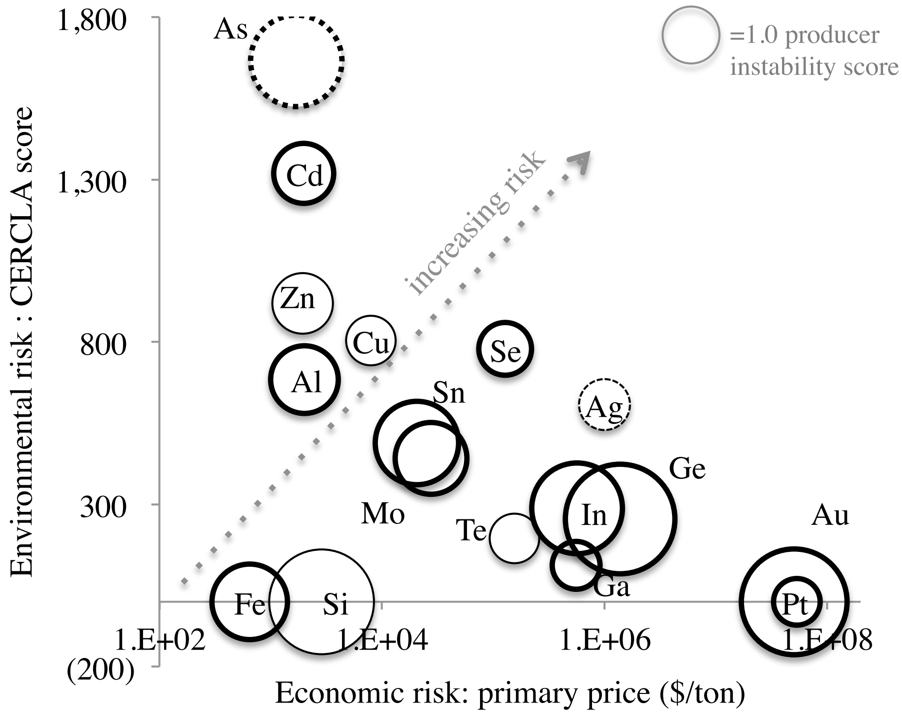


Figure 2.3 Relative criticality of PV materials using CERCLA score, primary socio-political stability indicators

This order of criticality is more pronounced in Fig. 2.4 where, the y-axis is primary embodied energy, and the size of the data point represents the ratio of reserves to production. Given these axis, Au, Pt, In, Ga, Ag are the most critical while again Fe is the least. For this set of metrics the risk increases with respect to price and primary energy but decreases with respect to the ratio of reserves to production. These relationships show that in general, energy intensive materials are at greater risk for depletion and are more expensive. As previously stated, the greater the primary energy, the greater the energy savings from secondary production. These relationships suggest that strategies aimed at increasing recycling may work to simultaneously address physical scarcity and energy consumption for economically valuable materials markets. Alternatively, strategies that seek to open new mine reserves could decrease price while increasing negative human health and environmental impact of critical materials.

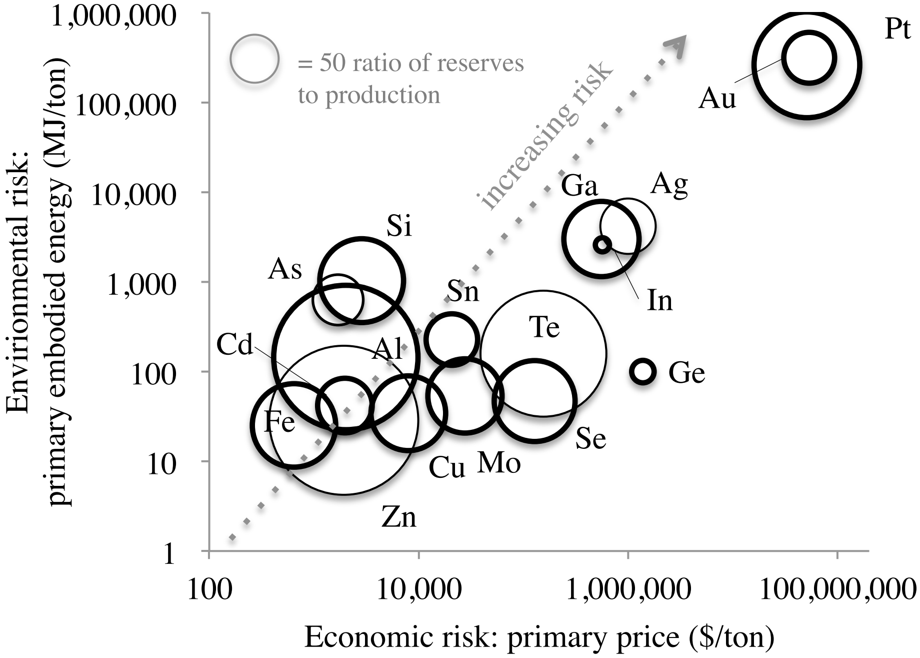


Figure 2.4 Relative criticality of PV materials using primary embodied energy, primary price, and the ratio of reserves to production indicators

For Fig. 2.5 increasing institutional and physical scarcity risk is towards the top right corner of the figure; the increasing size of the data point represents increasing environmental risk or increasing CERCLA score. Therefore, As, Te, Ga, Ge, and In are the most critical while Fe is the least. For this set of metrics, institutional inefficiency and environmental risk are weakly related for some PV materials. In general, as import reliance increases, the toxicity risk and frequency at Superfund sites decreases. This relationship suggests that domestic environmental risks related to mining are being diminishes as our reliance increases. The global impacts of this activity include greater environmental i.e. climate change and human health problems that are being shifted to other parts of the (developing) world where there is less stringent environmental standards and lower technology efficiency. Policymakers can utilize this set of metrics to develop comprehensive strategies that promote secondary production, domestic mining, or investment in the efficiency and environmental safety of foreign mining operations.

The relationships observed from Fig. 2.5 demonstrate how aggregating an indicator may conceal the underlying reasons for relative risk rankings. In the case of supply risk determinations, some materials e.g. Ge, As, Si, In have acute producer political instability and issues whereas others have more import dependence issues e.g. In, Sn, Zn, Se, As or physical scarcity issues e.g. Au, Se, Te. If we were to aggregate these metrics the individual effects may cancel one another. Si and Pt demonstrate another interesting case in terms of criticality determinations: that the use of particular indicators may drop some materials off the list of concern. For example, Pt does not appear on at Superfund sites and therefore has no CERCLA score. Another example, Si is very abundant material such its reserve data is highly uncertain. Therefore these Pt and Si are difficult to accurately map relative to other materials using these metrics. Despite these challenges, this methodology can be applied to other countries where material specific supply, economic, and environmental data are known.

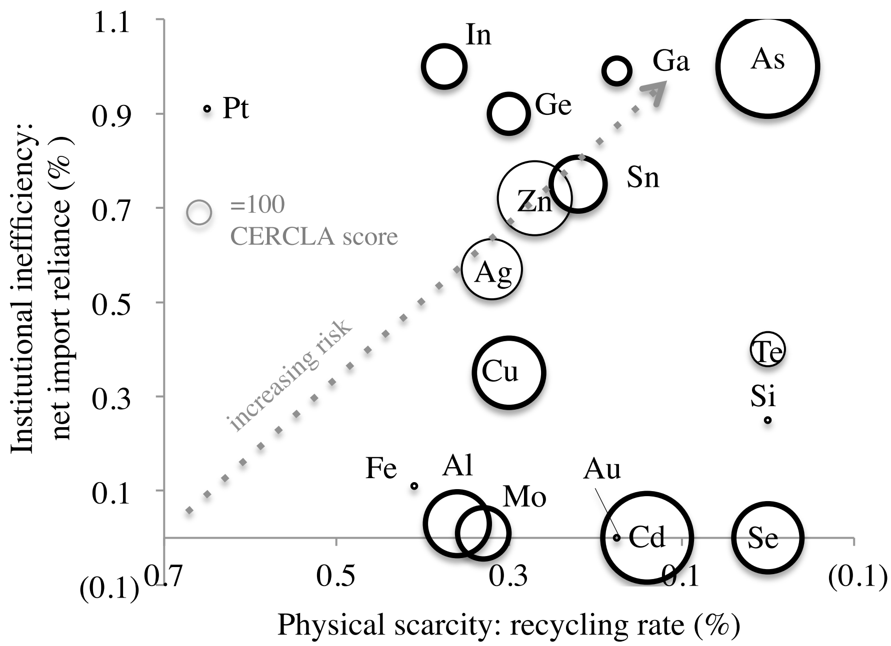


Figure 2.5 Relative criticality of PV materials using net import reliance, recycling rate, and CERCLA score

2.4 Conclusions

Our analysis determined PV materials in relative order of most to least critical for these metrics are: Ge, Pt, As, In, Sn, Ag, Se, Si, Te, Cd, Zn Au, Ga, Cu, Mo, Al, and Fe. Of these, Se, Fe, Cd, Ag and Zn are the most important materials to the economy in terms of the number and size of industrial sector applications. In terms of consumption, less expensive materials e.g. Fe, Al have greater value due either to larger volumes. Compared to best and worst case scenarios, nearly all of the PV materials studied have high producer political instability due to the high concentration of production among one or two producers. When PV materials are compared to more common materials such as Fe, they all exhibit greater energy intensity and most are present at Superfund sites. Multi-metric analysis reveals tradeoffs that suggest friction between sustainable economics, political stability of supply, and environmental quality objectives. We have proposed moderate, long-term policies e.g. education, subsidies, information and aggressive, short term polices e.g. expanding defense, financing, regulation, and taxes aimed at delaying or mitigating criticality issues of PV materials. Moderate policies may require coordination between federal and state governments whereas aggressive policies are more confrontational, possibly sparking further conflict with adversarial nations in this realm.

There are questions remaining in this area, particularly around future criticality, policy, and resource management. Current trends indicate that many countries are moving towards aggressive actions to secure resources necessary for economic growth and infrastructure development while material prices, and consumption rates increase. These factors point to increasing conflicts over the appropriate and effective actions that may shield domestic markets from supply disruption. Future work in this area is also required to understand the global dynamics of criticality, for example, what is the impact of increasing populations and affluence of emerging economies on physical scarcity and the environment? What politically feasible policies can decouple economic growth, resource depletion, and environmental impacts to mitigate future criticality risk? In addition, several policy interventions have been proposed here that could be explored further such as whether encouraging recycling could mitigate criticality or if the US should invest in domestic mining to ensure future materials availability? Additionally, several previous studies have joined us in proposing the use of substitutes, however, the supply, environmental, and socio-political tradeoffs of the proposed substitutes is unclear.

3. STRENGTHENING THE CASE FOR RECYCLING PHOTOVOLTAICS: AN ENERGY PAYBACK ANALYSIS

3.1 INTRODUCTION

Renewable energy technologies i.e. hydro, biomass, and solar have emerged to address the negative environmental impacts of increasing use of fossil fuels. Solar photovoltaics (PV) are an attractive renewable energy technology because they avoid significant carbon emissions during the use phase common to non-renewables, have a long useful lifetime estimated at 20 – 30 years, and they take advantage of a stable and plentiful energy resource – the sun. In PV research and development, there is a strong emphasis on efficiency gains as one of the best strategies to increase the technology's economic and environmental attractiveness.[1] However, efficiency, while important, is only one strategy for reducing environmental impact and increasing energy savings. Recycling is another strategy with potential that has yet to be fully recognized due to the current lack of collection infrastructure and uncertain set of processing technologies. The use of secondary materials in production has the potential to reduce material energy intensity as well as improve economics by providing a less expensive material supply. This work explores under what conditions energy payback from increases in recycling is equivalent to increases in efficiency.

For a significant number of primary and secondary PV materials LCA data is either incomplete or unavailable, for this reason we use cumulative energy demand data to evaluate energy payback. Energy payback time (EPBT) is the energy analogy to financial payback, it quantifies the time it takes for the energy produced after technology installation (in terms of primary energy equivalent) to equal the total energy required to produce it (including the energy burden of materials, manufacturing, collection, and disposal). For example, when solar PV technologies generate power they offset the energy spent to harvest the materials used in their production and manufacturing. Increasing efficiency improves EPBT by increasing energy generation. Alternatively, increasing recycling reduces life-cycle energy spent to harvest and refine PV

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[1] For example, the US Department of Energy DE-FOA-0000492 Foundational Program to Advance Cell Efficiency awarded over $19 million to research projects to advance PV efficiency

materials. Previous literature has widely used energy and CO2 payback to quantify the environmental impacts of energy systems[88-93].

PV materials, especially those used in the absorber layer (e.g. Si, Te, Ge, In), consist of metals that have high cumulative primary energy demand compared to most materials with the exclusion of precious metals (e.g. Pt, Au). Another factor that increases the energy burden of PV materials is the refining necessary to achieve a minimum purity required for performance. For example, the Siemens process used to refine silicon into semiconductor feedstock used for PV is of up to 99.9999% purity and estimated to account for 75% of a polycrystalline silicon (c-Si) PV module's total production energy [94]. Similarly CdTe semiconductor material for PV is assumed to be between five and six 9 purity in many life-cycle assessment (LCA) studies [95]. In addition to the high purity, some PV materials reflect a high processing energy because they are produced in low concentrations as a by-product of other mining such as Cu or Zn (e.g. Te, In, Ga, Ge) or require energy intensive production techniques such as electrolysis (e.g. Al production from bauxite). Because recycled materials require significantly less processing and refining compared to primary materials, the potential energy savings is significant for PV materials. On the other hand, as compared to bulk materials, the purity requirement for cell materials makes recycling more demanding in terms of cost and energy input.

While this work focuses on quantifying the energy savings potential through recycling, using secondary materials has other benefits for PV technology as well. One is the potential to significantly reduce costs; while scrap metals follow their primary commodity price, there is typically a discount of 10-80% depending on the scrap quality [96]. In addition, the use of secondary materials contributes to waste reduction by diverting materials from landfills and back into the market. A well-developed secondary material infrastructure also has the potential to mitigate scarcity issues [87]. Recent work has highlighted resource scarcity and criticality as a potential issue for PV materials like In, Ga, and Te [37],[36]. While the research community is divided on how severe this issue may be, all can agree that future supply has a great deal of uncertainty due to a variety of factors including PV adoption, recycling policy, majority mine ownership and management, electronics demand, and price. Although future demand of PV will likely rapidly outpace supply from secondary sources, such potential energy, cost, and scarcity mitigation would still be significant for high utilization.

The PV technologies analyzed in this study each have unique processing, composition, and properties. Silicon-based technologies – i.e. polycrystalline and mono-crystalline silicon are the most mature, one of the least expensive, and have one of the highest production efficiencies thus holding over 80% of the current market share. Thin-film technologies - i.e. cadmium telluride (CdTe), copper indium gallium diselenide (CIGS) and amorphous silicon (a-Si) - are named for their semiconductor layer thickness of just a few micrometers. Thin-films generally have more flexible applications due to their smaller size, lower efficiencies, and ease of manufacturing as compared to traditional silicon-based PV. Emerging technologies such as organic PV, dye-sensitized, and multi-junction PV are still in development; they have the widest array of material compositions and therefore are not analyzed here. This analysis focuses on the most mature PV technologies: silicon-based and thin films.

Previous work suggests that recycling processes for silicon-based and thin-film PVs at end-of-life are technically possible [80,97-99], have economic benefits [78], and have significant contributions to reducing the life cycle impact [100,101]. Furthermore, literature also suggests that the recycling of the module frame [102], recycling silicon wafers for c-Si[94], and the recycling of Ag and Zn in transparent conductive oxides [103] has a significant impact on energy payback time. However, a comprehensive accounting for recycling's impact of all direct PV materials in the energy payback calculation has not been performed. This quantification would allow a fair comparison between developing recycling technologies and efficiency gains as strategies to reduce the environmental impact of solar technology. This study explores the impact of recycled content on the energy payback time of silicon-based and thin-film PV modules. The energy payback time (EPBT) of PV modules containing recycled materials is evaluated to show in which regimes improvements in recycling rates can demonstrate equivalent energy savings to improvements in efficiency. In this effort we systematically compare silicon-based (i.e. c-Si) and thin-film (i.e. CIGS, CdTe, a-Si) PV technologies. Sensitivity of results to changes in module lifetime, composition, recycling rate, and configuration (i.e. ground-mounted, roof-mounted) are also investigated.

3.2 METHODOLOGY

3.2.1 Energy Payback Calculation

Energy payback is the ratio of energy input, E_I to energy output rate, $\dot{E_O}$ (Equation 3.1). The energy input to produce and manufacture each material, n, is determined by the cumulative primary energy demand, E_P, secondary energy, E_S, the composition, c, and recycling rate, r. The energy output was calculated using the solar insolation, H, performance ratio, PR, and a module efficiency, η. We assume a solar insolation of 1700 kWh/m^2/yr – i.e. average solar radiation in southwest U.S. and Spain - and system performance ratio for all technologies between 0.75 – 0.80 similar to [8,100,104-106]. However these results may vary with array orientation, tilt, and grid efficiency [91].

$$\boldsymbol{EPBT = \frac{E_I}{\dot{E_O}} = \frac{\sum_n c(1-r)(E_P) + r(E_S)}{PR\,\eta\,H}}$$

Equation 3.1

This way of describing energy payback is consistent with suggested LCA guidelines [107] which assumes that all of the manufacturing and production energy is primary (in the case of no recycling or $r = 0$) however we deviate from this assumption with the inclusion of a recycling rate and the secondary energy required to recycle PV materials. By using primary and secondary material cumulative energy demand for the energy input we explicitly include extraction, refining, production and recycling energy and omit operation, maintenance, assembly, end-of life transport, and indirect material use. We also deviate with suggested LCA guidelines by neglecting transmission and distribution losses from the grid which vary significantly by location. Typically the system components – e.g. frame, roof or ground mounting supports, inverter, and cables – are included separately from the PV cell however in this analysis we define the module to include the frame, mounting array supports, interconnects and the PV cell. For lifetime, based on data from literature [108], CdTe, CIGS, a-Si, and c-Si technology degradation rates do not vary significantly and are consistent with 20 – 25 year product guarantees of power from major cell and module manufacturers. However literature suggests the mounting frame could have a lifetime three times that of the module [102].

3.2.2 Material Composition

This energy payback analysis includes all direct materials on a mass (kg) per module area (m^2) basis assuming a baseline configuration (Table 3.1). The module configuration and associated compositions (Table 3.3) were determined from manufacturer technical specifications and PV literature. These compositions were chosen because of the relevance of their associated efficiencies. For example, a CdTe module efficiency greater than 9% likely represents the majority of currently installed modules since more than 80% were produced in the last 5 years[109].We recognize that for each PV technology there are multiple configurations and products with varying compositions; the impact of these differences within the observed range is reported in Table 3.1. However, similarities do exist between the technologies. In particular, each has three to four layers sandwiched between a top layer of glass and a substrate, either 3 – 4 mm thick glass or 0.1 - 0.2 mm stainless steel. The ethyl vinyl acetate (EVA) encapsulant is an adhesive between several layers including the back contact and substrate is not included in this analysis. However the anti-reflective coatings such as MgF_2 of 0.1 μm thickness – included in Table 3.3 - are included.

In order to find the state of the art efficiency of various thin-film and silicon-based photovoltaic modules, manufacturer technical data for framed and frameless modules, was used. The frameless CdTe modules (n=13) in this dataset were all produced by FirstSolar. The CIGS modules were produced by Q cells (n=6), SunshinePV(n=1), Solibro Gmbh (n=11), Nanosolar (n=6), Avancis GmbH & Co KG (n=2). The a-Si modules (n=18) in this data set were produced by two US based manufacturers Xunlight (n=7) and United Solar Oovonic LLC (n=11). The c-Si modules were produced by Yingli (n=4), JA Solar (n=3), Trina Solar (n=3) and Suntech (n=3)[13,110,111].

Table 3.1 Baseline PV Cell Configuration and Layer Thickness in μm

Layer (Thickness)	*CIGS*	*CdTe*	*a-Si*	*c-Si*
Efficiency [13,110,111]	13.5	14	8.2	20
TCO/Contacts	Al:ZnO/ ITO (0.07 – 0.25) [100,104,105,112,113]	SnO2Cd2SnO4/ ITO (0.2 – 0.5)	ITO (2.0)	Ag (0.02)

		[80,114]		
Window	CdS (0.05 – 0.07) [113]	CdS (0.6 – 2.0) [80,115]	-	-
Absorber	Cu,In,Ga,Se/S (1.0 – 3.0) [80,112,113,115]	CdTe (2.0 - 4.0) [115,116]	a-Si:Ge (0.8) [117]	Si (300 – 400) [104,116]
Back Contact	Mo (0.5 – 1.0) [80,113]	ZnTe:Cu/Ti (0.5) [118]	Al/ZnO (0.15)	Al (0.01)

As previously stated the material compositions of frame and mounting materials are included in this study with assumptions for the baseline case as shown in Table 3.2. Roof and ground mounts vary in design and purpose – i.e. residential or distributed power installations - however the industry has converged on the use of steel and aluminum for array supports and rails. In addition, more recently developed mounts accommodate both framed – i.e. a-Si, CIGS - and frameless – i.e. CdTe - thin film modules. For example, IronRidge and Schuttler produce residential flat roof mounts that clamp to a frame or use adhesive to attach to a back contact (frameless modules) and slide onto rails nailed to a roof. These mounts can also be used for distributed power installations, which require pile-driven or concrete secured steel posts in the ground for rails to attach. Therefore we use technical data from IronRidge (roof and ground mount products) to develop a range of Al and Fe compositions for flat roof, slanted residential roof, and distributed power ground mounting schemes. These assumptions are within the ranges of previous studies [102,119,120].

Table 3.2 Array Mounting and Frame Baseline Composition Mass per Area (g/m^2)

Description	*Al*	*Fe*	*Cu*
Slanted roof mount & interconnects	1708	197	27.8
Ground mount & interconnects	1708	2029 -7682	27.8
Frame only	1500		

Table 3.3 PV Cell Baseline Composition Mass per Area (g/m^2)

CIGS	*CdTe*	*a-Si*	*c-Si*
Al (< 0.1)	Cd (13.1)	Al (0.2)	Ag (0.1)
Cd (0.2)	Cu (0.7)	Fe (600)	Al (270)
Cu (2.2)	Glass (13,380)	Ge (1.9)	Glass (6,690)
Ga (2.4)	Sn (1.0)	Glass (6,690)	Si (698.7)
Glass (13,380)	Te (10.9)	In (0.4)	Mg (0.1)
In (7.7)	Zn (0.8)	Si (4.6)	
Mo (4.3)		Zn (0.5)	
Se (2.7)			
Sn (0.4)			
Zn (< 0.1)			

3.2.3 Secondary Energy Estimation

This EPBT analysis includes primary and secondary cumulative energy demand data obtained from SimaPro 7.3 using the Ecoinvent 2.2 database according to [57]. Although it is key to energy analyses, secondary energy demand data is not available for a variety of materials within this database, mainly due to the immaturity of recycling technologies for specific metals. This includes several PV materials: Si, Se, Cd, Zn, Sb, Te, In, Ga, and Mo. Unknown secondary energy was estimated assuming that energy savings (the difference between primary and secondary cumulative energy demand) scale proportionally with primary energy; this assumption is based on a clear linear trend seen for available data which includes Au, Ag, Ni, Fe, Al, and Cu. The linear trend (Equation 2.2) for available data was regressed and coefficients *a* and *B* were found to be 0.9762 and 16.361, respectively, with high correlation to existing data (R^2 =1). Missing secondary energy data was then estimated using this regression relationship and known primary energy data as shown in Fig. 3.1.

Although in reality, there may be some deviation from extrapolation due to particular processing technologies, impurity of the secondary materials, or geographic location of processors; we found this to closely approximate previous literature on the recycling energy requirements for end-of-life silicon wafers [13,94,110,111] and CdTe recycling for thin-film technologies[104,121,122].

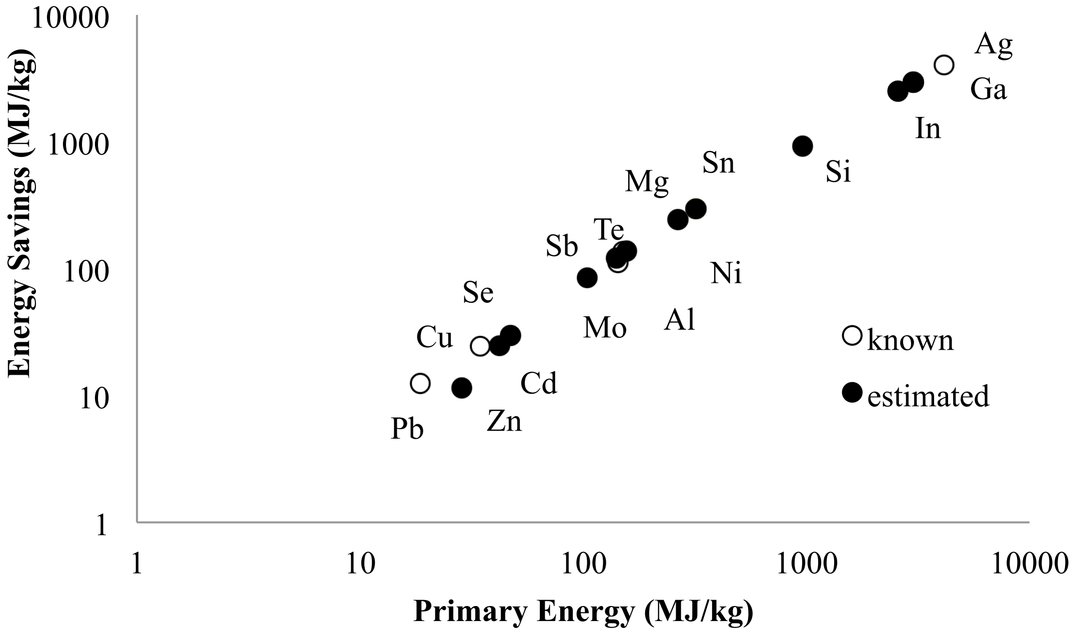

Figure 3.1 Estimated energy savings of PV materials on log-log scale

3.2.4 Recycling Rate

Recycling of end-of-life PV modules is currently negligible due to a variety of factors discussed above. In order to explore the potential energy payback savings provided by recycling, both an overall recycling rate of modules must be assumed as well as individual material recycling rates. The United States Geological Survey (USGS) municipal solid waste recycling rates for individual materials contained in PV, abbreviated as MSW RR, were used for this EPBT analysis as listed in Table 3.4. USGS defines the recycling rate as the supply fraction that is scrap, on an annual basis. The recycling rate is equal to the sum of consumed old scrap and consumed new scrap divided by the sum of apparent supply, imports, exports, and adjustment for government and industry stock change as shown in equation 3.2. One important distinction is the difference between “old” and “new” scrap. “Old” scrap is collected from discarded or end-of-life products while “new” scrap (also called “prompt” or “run-around”) is generated during fabrication and manufacturing. While both types of recycling offset primary production, recycling end-of-life materials or old scrap is considerably more compositionally challenging[123]. Apparent supply is the sum of primary and secondary production. This calculation of recycling rate includes products made in current and prior years as well as new scrap not reused in the plant, as shown in Fig. 3.2. Expended or obsolete material unable to be recycled due to dissipative uses is not included in this recycling rate calculation. The USGS and United Nations Environment Programme (UNEP) municipal solid waste recycling rates shown in Table 4 contain uncertainty due to many factors such as varying lifetimes of products, amounts dissipated, and slag emission losses.

$$\text{MSW RR} = \frac{\text{consumed old scrap} + \text{consumed new scrap}}{\text{apparent supply} + \text{imports} - \text{exports} + \text{adjustment}}$$

Equation 3.2

Table 3.4 MSW Recycling Rates (%) for PV Materials

Material	*MSW RR*	*Material*	*MSW RR*	*Material*	*MSW RR*
Al	36%	Ge	30%	Se	0%
Ag	32%	Glass	20%	Si	0%
Cd	14%	In	< 1%	Sn	22%
Cu	30%	Mg	33%	Te	0%
Fe	41%	Mo	33%	Ti	52%
Ga	18%	Ni	41%	Zn	27%

Source: [62,124,125]

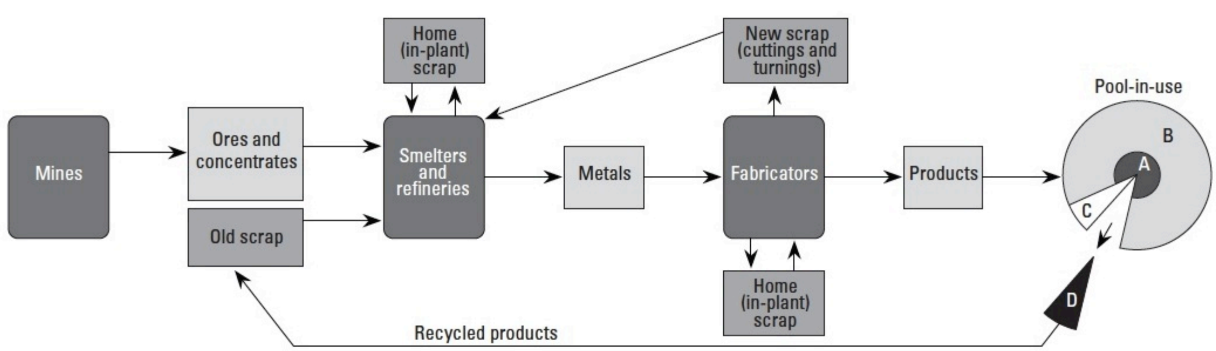

Figure 3.2 Generalized metals recycling flow chart that demonstrates how products made in current year (A), products made in prior years (B), unrecovered products (C), and recycled products (D) are related for end-of-life products [124]

3.3 DISCUSSION

3.3.1 Material Energy Intensity and Value

The material primary embodied energy and material value of mature silicon-based and thin-film PV is dominated by the frame and mounting materials. The mounting components contribute 7 - 13 MJ/metric ton (mt) primary embodied energy and \$1600 – 2300 per mt primary material value in the baseline case. This is further demonstrated by the frameless CdTe design which has the lowest embodied energy and primary material value as compared to all other framed module designs. When looking at framed thin-film modules, without mounting, as shown in Fig. 3.3 and Fig. 3.4, we see that although substrate materials i.e. glass, Fe are the second greatest contributors to material primary embodied energy they retain a smaller portion of primary material value in \$/mt module. The opposite is true for absorber materials e.g. In, Te, Si whose combined primary material value is a greater portion of the total per ton module than the material primary embodied energy for each technology. Interestingly, for thin-film technologies, when the substrate material is stainless steel as in the case of a-Si, the substrate has a greater material value but smaller portion of embodied energy.

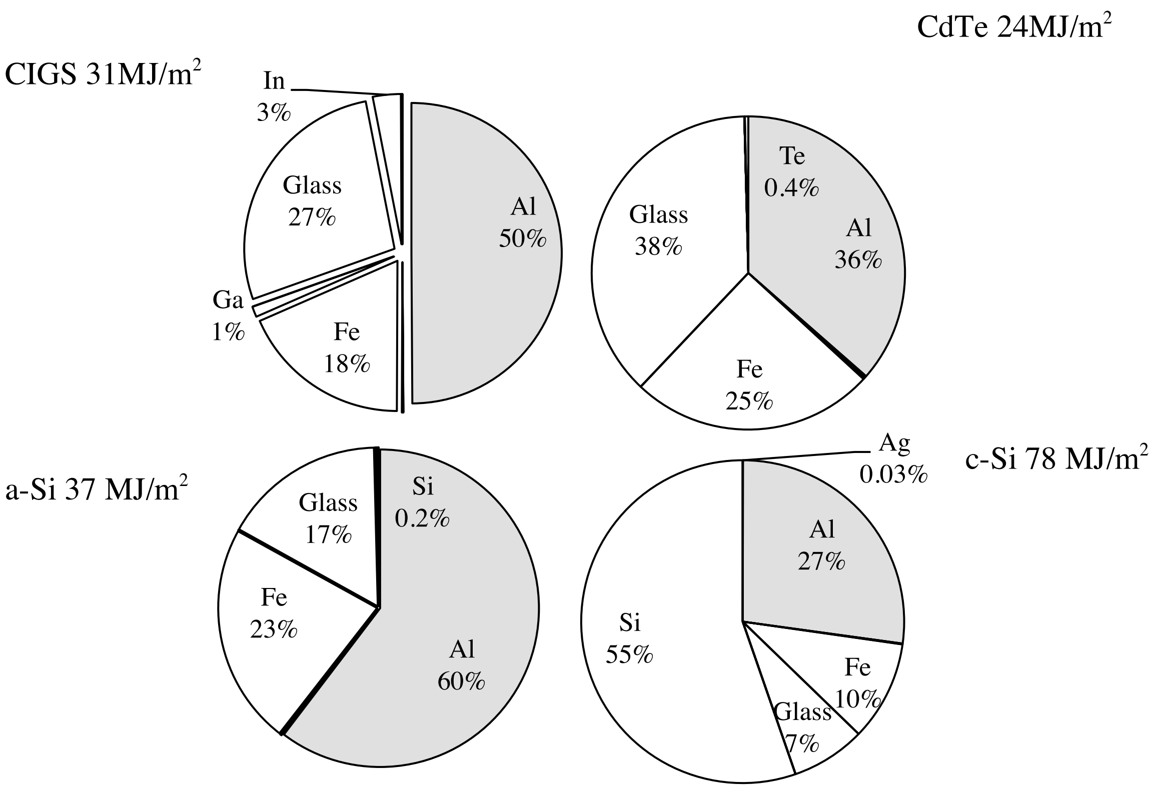

Figure 3.3 Primary embodied energy of 1 scrap kg of material for each PV technology (ground-mounted). Materials that do not appear on the figure have values of less than 0.01%.

For framed silicon-based modules, the absorber material i.e. Si dominates the primary embodied energy and Ag metal contact material dominates material value while frame materials are the second greatest portion for both metrics. The use of a stainless steel substrate in place of glass and Ag in place of ITO both increase the material value and embodied energy intensity of mature silicon-based PV above that of thin-film technologies.

These results have several implications for PV module design and recycling incentives. In order to encourage recycling, frame and mount systems can incorporate design for disassembly principles by eliminating cement structure in mounting arrays, using fewer types of fasteners, or non-proprietary fasteners. Towards the same goal, cell manufacturers can incorporate design for recycling by using encapsulation materials that are not harmful when vaporized or that melt at low temperatures without degrading substrate or absorber materials. We have shown that absorber materials have some potential economic value at end-of-life which previous PV economic studies [78,126] show for thin-film technologies depend on the price of absorber materials. For example, Te is recycled for CdTe due to its economic value, however, from an EPBT perspective, Te has low priority for recycling compared to frame and array materials. However, these studies do not include frame and mounting materials whose end-of-life material value provides further economic incentives to recycling. Other PV materials such as Ag, Al, and Fe contained in mature silicon-based modules have high or rapidly increasing prices as compared to potential substitutes i.e. Cu, glass which can also impact recycling incentives. The choice of materials is also important for the ease of recycling. In general, new scrap has a higher purity and is recycled more easily than post consumer or old scrap. For this reason, new scrap has less comingling and potentially higher material value. Cell designs that increase ease of recycling can reap similar material value and purity benefits for example see [127]. These results demonstrate that for all PV technologies, substrate, frame, and absorber material choices can impact the end-of-life material value and embodied energy thereby influencing incentives for recycling.

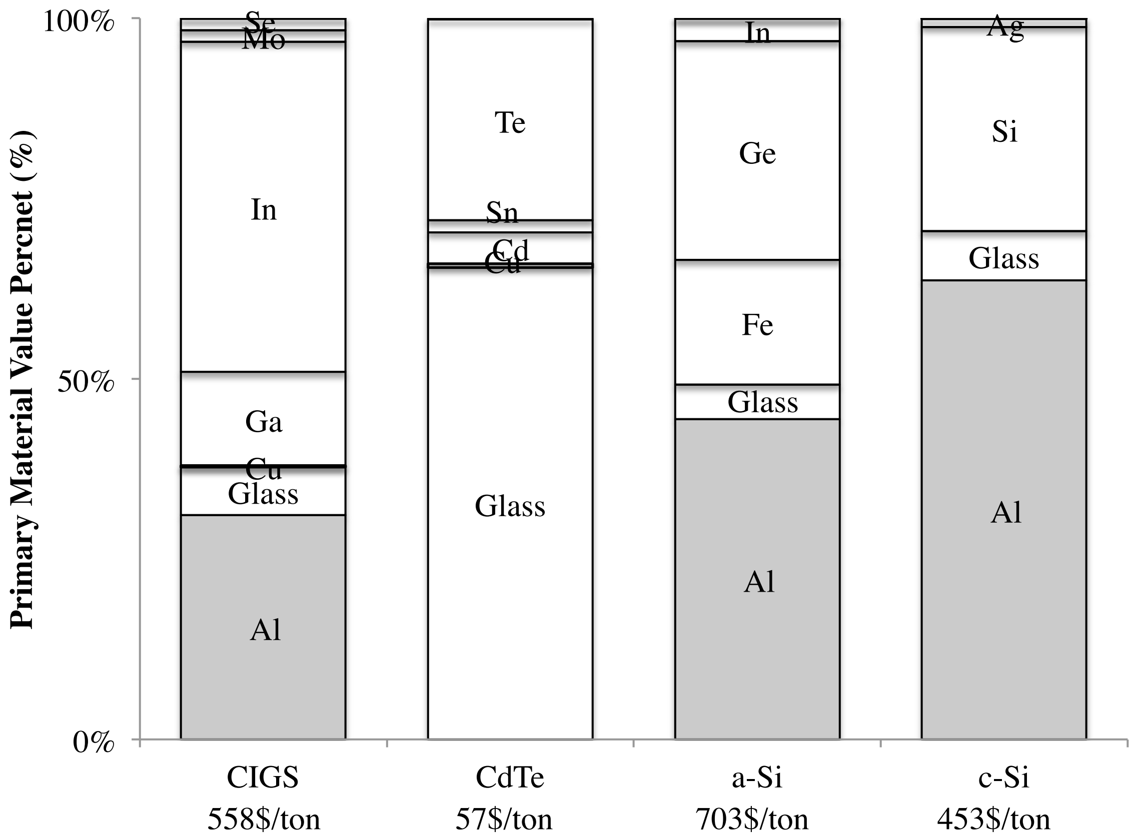

Figure 3.4 Potential material value for framed modules without mounting. Materials that do not appear on the figure have values of less than 1%.

3.3.2 Baseline EPBT

The EPBT of thin-film modules decreases linearly as material recycling increases for a given efficiency. The lower the module efficiency the steeper the decline of EPBT with recycling rate for all technologies (Fig. 4). Exhaustive recycling, ER, of all materials reduces EPBT by 0.5, 0.7, 1.1 and 1.1 years for CdTe, CIGS, a-Si and c-Si at baseline efficiency. With no recycling (NR) the EPBT is 0.8, 1.0, 1.5 and 1.3 years for CdTe, CIGS, a-Si and c-Si PV modules respectively at baseline efficiency. Recycling all materials at their respective municipal recycling rates (MSW) reduces EPBT by 0.2, 0.2, 0.5 and 0.2 years for CdTe, CIGS, a-Si and c-Si at baseline efficiency. These EPBT values are comparable to previous studies that include limited or no recycling assumptions for thin-film and silicon-based PV technologies [102,105,128,129].The results imply that there is greater potential for embodied energy reduction from recycling lower efficiency modules.

There is also a possibility that thin-film technologies, due to lower efficiency and greater flexibility, will be incorporated into consumer power applications with shorter lifetimes and more geographic dispersion for example handheld electronics. Ironically, although retaining more energy saving potential, low efficiency PV powered electronics applications maybe less likely to be recycled as compared to larger grid connected systems due to smaller size, wider geographic dispersion, and the absence of frame and mount materials. Geographic dispersion is also a concern for end-of-life collection of PV systems and there are some take-back models proposed in the literature [130] [79]that address this concern. Several previous LCA studies have emphasized that transportation can influence the sustainability of end-of-life options [131-133]. Particularly in the case of electronics where there are small quantities of many different materials of high value where local recovery facilities may not have the processing capability to recover old scrap so that its composition/purity matches that of new scrap. Transport is also not included in secondary cumulative energy demand LCI material data. Our results add to this discussion by prioritizing technologies and components for recovery from an energy payback potential Fig. 3.4 and material value perspective Fig. 3.3. Applying these results to PV policy has implications for adoption and end-of-life policy. Current U.S. PV policy is focused on adoption incentives e.g. tariffs, subsidies without regard to recovery or end-of-life collection. Considering potential future PV applications, material value, and potential energy savings PV policy will need to weigh additional criteria such as geographic dispersion of PV installations, technology generation, expected lifetime, and module efficiency in order to develop environmentally sustainable PV adoption and end-of-life policy. For example, if recycling increases by mandate or other mechanisms then material recovery facilities must prepare technologically and logistically to handle the material flows of end-of-life PV modules. Work that focuses on PV recycling economics in the context of U.S. policy and waste infrastructure is important (especially as a context to this EPBT analysis) but not well characterized and is therefore in progress by the authors.

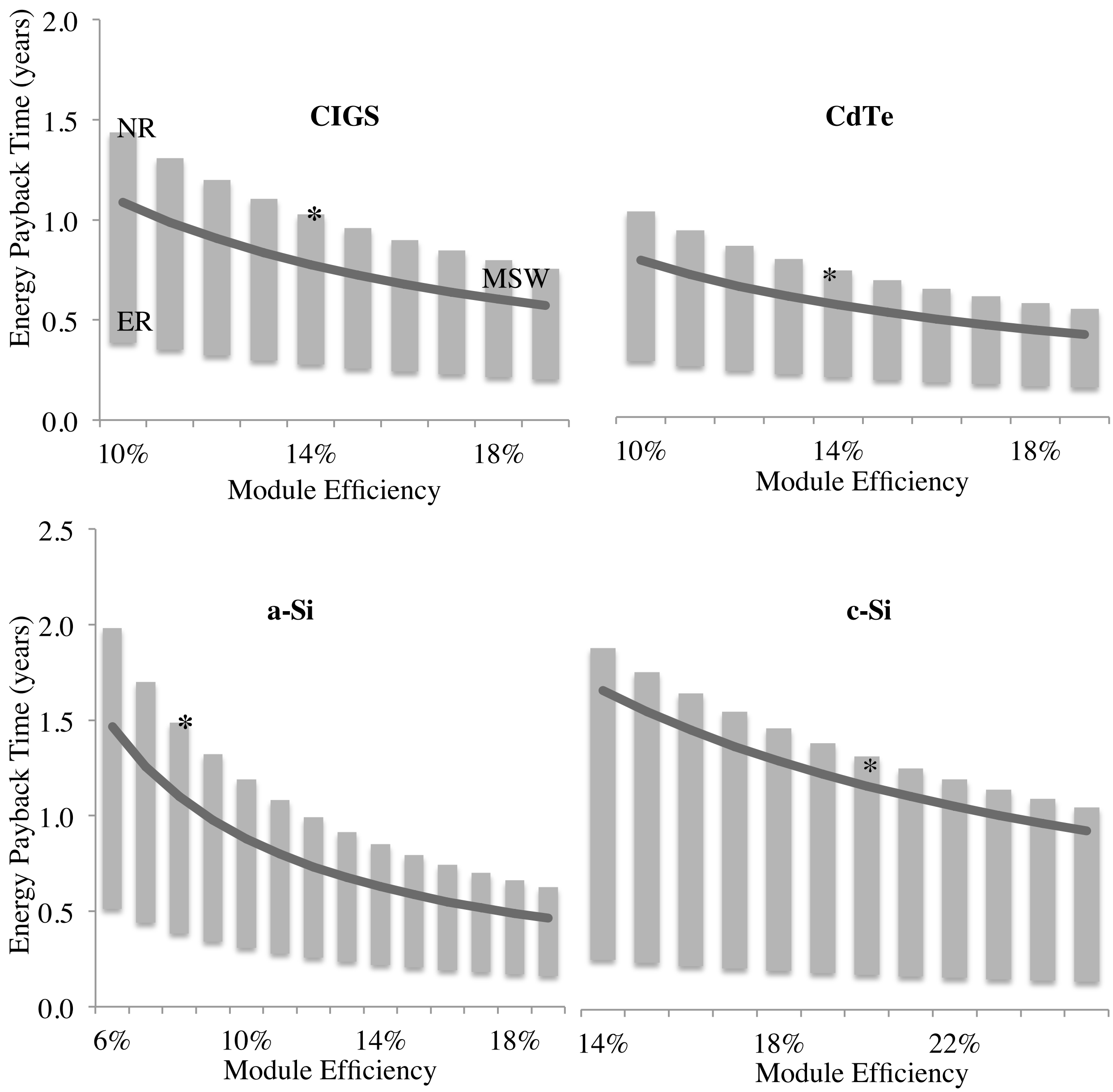

Figure 3.5 EPBT of CdTe, CIGS, a-Si and c-Si ground-mounted PV modules with varying module efficiency for exhaustive recycling (ER) to no recycling (NR), and municipal solid waste (MSW) recycling rates scenarios. Asterisks (*) indicate a current base case that reflects today's installed average efficiency and negligible recycling

3.3.3 Component-level EPBT

For thin-film technologies, the reduction in EPBT from increasing the recycling rate is disproportionally influenced by the material with greatest primary energy demand per area: the aluminum frame and roof mounting rails as shown in Fig. 3.6. Applying the MSW recycling rate to frame and roof mounting materials has the potential to reduce EPBT by 0.2 - 0.5 years. For mature silicon-based technologies, the cell materials disproportionally influences EPBT. Applying the MSW recycling rate to cell materials e.g. In, Te, Si has the smallest potential reduction on EPBT of less than 0.1 years. MSW material recycling rates is one indicator of the relative intensity of secondary material use and as demand stabilizes can indicate progress toward reductions in primary metal. The substitution of secondary materials has the potential to reduce energy use, resources, water, and land use compared to extraction and processing of primary material.

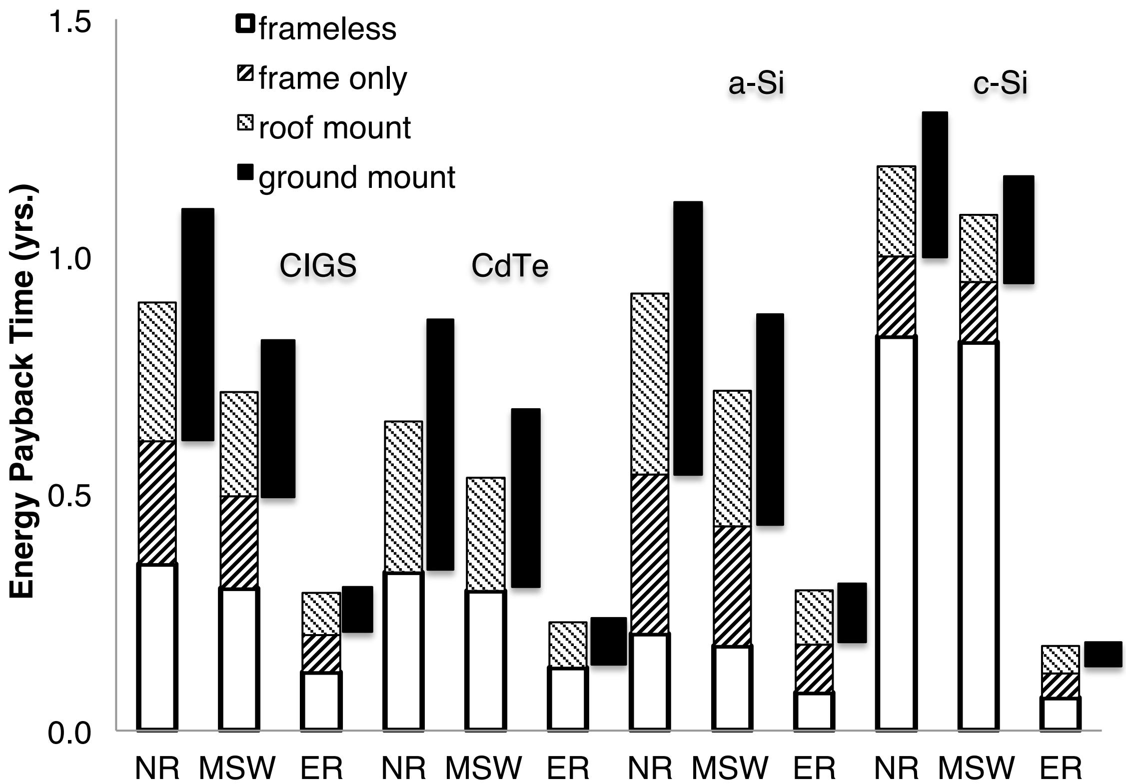

Figure 3.6 The contribution to EPBT of base case frameless module, frame, roof mount, and ground mount EPBT for no recycling (NR), municipal solid waste recycling rate (MSW), and exhaustive recycling (ER) scenarios for each technology.

3.3.4 Efficiency vs. Recycling Rate

The EPBT of PV modules decrease as efficiency increases for a given recycling rate. This means the impact of recycling on EPBT decreases as the efficiency increases. This implies that greater potential for embodied energy reduction from recycling for low efficiency modules, In general, a 3-5% change in the recycling rate produces a reduction in EPBT equivalent to a 1% change in efficiency. For example, a 100 day reduction in EPBT (from 200 to 100 days) can be achieved equally by a 30-50% improvement in recycling rate (from 66% to 100%) for all materials or a 10-12% efficiency improvement (from 12% to 22%) for CIGS modules as shown in Fig. 3.7. These effects are non-linear and also are housed within the context of relative effort to achieve these gains. For example, an efficiency gain of 1% may require significant effort (i.e. design cost, research and development, etc.) compared to the effort required for an equivalent increase in recycling rate (i.e. enhanced collection infrastructure, processing technologies. The longer the lifetime of the PV module the smaller the primary material energy avoided. While an EPBT analysis does not take in to account lifetime, it does provide a number to compare to rapidly changing lifetimes.

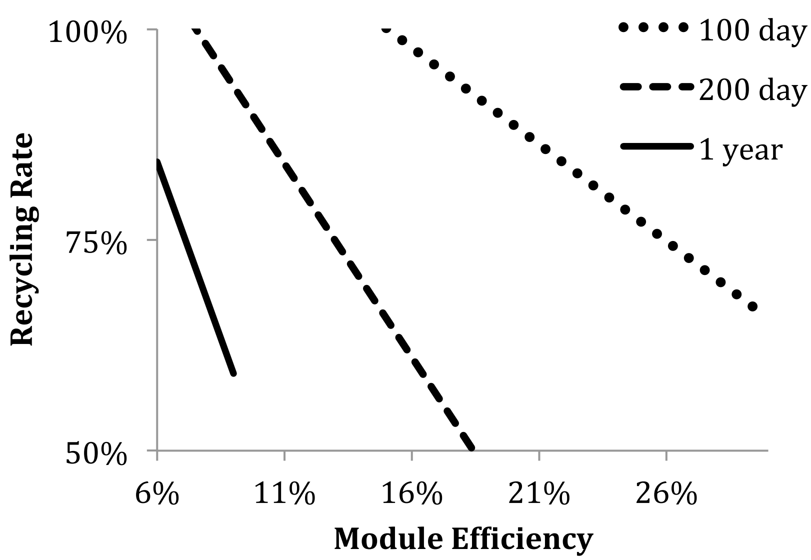

Figure 3.7 Material recycling rate and module efficiency tradeoff line for an EPBT of 100, 200 days and 1 year for CIGS modules

3.3.5 Material-specific Issues

The previous results showed reductions in EPBT assuming that all of the materials contained in PV recycling rates were being increased by the same percentage. In reality, it is more likely that secondary processors will target particular materials, as some are much easier to extract and recover than others. Materials like aluminum, copper, and steel have robust recycling technologies and infrastructure already in place while more PV specific materials such as indium, gallium, tellurium, and silicon do not. The three PV materials with the greatest energy savings per area and therefore likely recycling targets from an environmental impact perspective are the Al frame, glass substrate, and In absorber materials for CIGS modules. Research and some production level PV designs have proposed the elimination all or most of these high-impact materials with frameless designs, new absorber compositions e.g. CZTSSe, and polymer substrates. These design changes will likely shift the focus of environmental impact on array supports and energy intensive cell materials. It is also likely that current composition with glass substrates will be difficult to recycle and need to be separated from the general glass waste stream due to contaminants introduced by anti-reflective coatings e.g. Mg. Recycling each of these materials yields substantial reductions in EPBT. The impact of recycling individual materials is non-linear similarly to the materials as a whole. For example, as the end-of-life (EoL) aluminum recycling rate increases, the EPBT decreases at a decreasing rate as shown in Fig. 3.8. This means that the influence of recycling aluminum is reduced as module efficiency increases, although still providing a significant net benefit. For example, a reduction in EPBT of 66% for increasing from no recycling to 100% Al frame recycling (for 6% efficiency). The EPBT benefits of recycling the frame at EoL are significant despite the assumption that the secondary material content of aluminum in the frame is equal to municipal solid waste recycling rate of 36%. A similar trend can be seen for increasing the recycling rate of the glass substrate although slightly less dramatic. For a cell with 6% efficiency, a 26% reduction in EPBT can be gained by increasing glass substrate recycling from zero to 100%. In contrast, due to its low mass contained within PV, a 100% increase in indium recycling provides only a 3% reduction in EPBT. Several studies have suggested the use of polymers e.g. polyimide [113] to replace glass substrate which may have the potential to reduce weight of current modules however, because there is gap in the LCI data it is unclear whether these alternate polymer substrates would also reduce embodied energy although previously assumed has not been the case for other PV technologies[134].

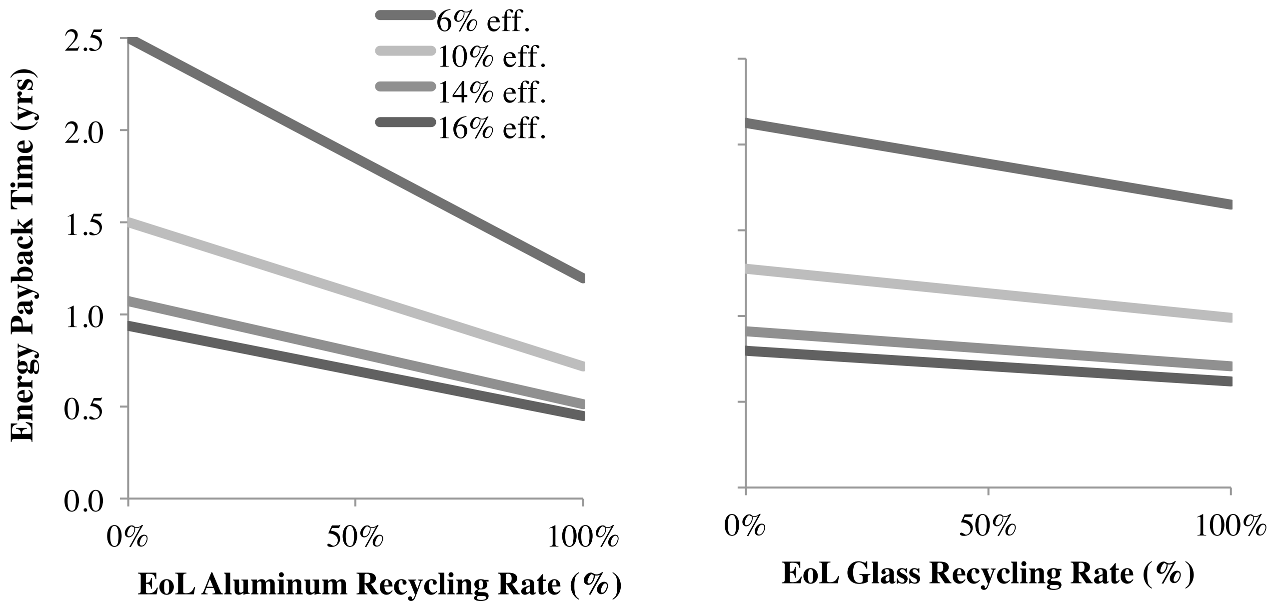

Figure 3.8 CIGS module energy payback time curve with varying aluminum frame, glass, and indium recycling rates and module efficiencies all other materials are assumed recycled at MSW rates

In order to evenly investigate the impact of both recycling rate and efficiency, we assumed modules contained the same structure despite efficiency increases similar to previous literature[128,135,136]. However, we recognize that efficiency improvements typically require different layer thicknesses, contact material and coatings. Therefore this assumption oversimplifies the compositional reality of sizable gains in cell and module efficiency. To address this simplification we compare the composition of three record efficiency modules from research literature [137-140] to extrapolations of our baseline composition in Fig. 3.9. The extrapolation of baseline CIGS overestimates the EPBT savings from exhaustive recycling and underestimates the EPBT in the case of no recycling for research compositions of CIGS at 18.8% and CZTSSe at 10.1% efficiency. Conversely, the extrapolation of baseline CdTe underestimates the EPBT savings from exhaustive recycling and overestimates the EPBT in the case of no recycling for the research composition of CdTe at 16.5% efficiency. The deviation of EPBT between the compositions with the same efficiency is most pronounced in the case of no recycling. However, EPBT deviates less in the case of exhaustive recycling. We believe this trend occurs because the research compositions employ a larger number of cell materials than the baseline. Also, as previously observed, the frame, glass substrate, and array support materials dominate EPBT so that the exhaustive recycling scenarios are similar. Overall, we expect the further EPBT is from the baseline composition, the greater the deviation from future high-efficiency, production scale compositions.

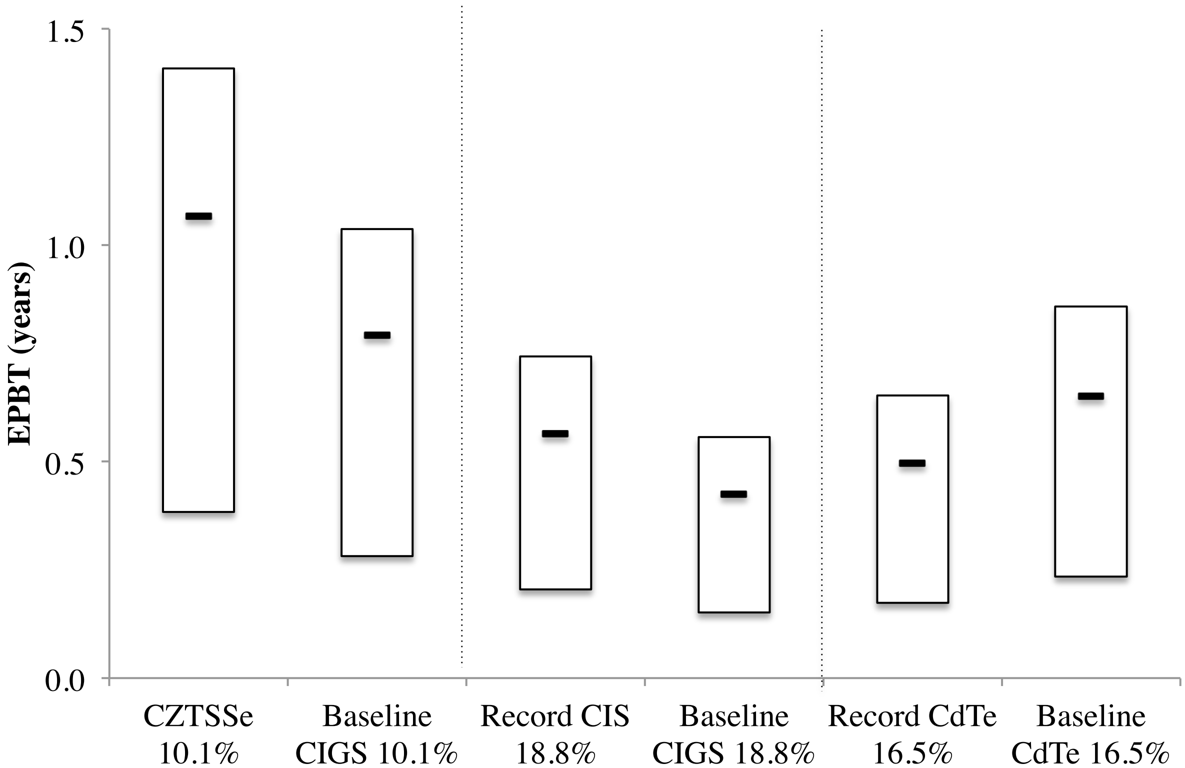

Figure 3.9 Comparison between extrapolated baseline compositions and record-efficiency research modules EPBT for CIGS and CdTe technologies.

3.4 CONCLUSIONS

Intuition would suggest that cheaper, low efficiency devices might be best thrown away, while expensive, high efficiency devices deserve attention to reuse and recycling options. However, these results show that from an energy payback perspective, the opposite incentive exists in terms of prioritizing recovery and recycling. This work aims to inform material and design choices to enable minimization of life-cycle energy embodied in PV technologies. Exhaustive recovery of PV materials has the potential of reducing energy payback time of mounted modules by more than half for mature silicon-based and thin-film technologies. The aluminum frame and mounting rails of the PV system is responsible for the majority of these EPBT reductions. The focus on embodied energy impact of frame materials means module, cell manufacturers, and frame installers have an opportunity to work cooperatively toward extending their supply chains to reduce costs by recycling frame, mount, and cell components. This course of action also requires PV adoption policy to use criteria beyond installed capacity e.g. geographic dispersion and recycling rate to determine their long-term environmental impact.

This work highlights several takeaways for the PV community in areas of design, end-of-life recovery, research and development. Frameless designs decrease system EPBT, cost and disassembly complexity – a recycling incentive. Compositions that incorporate multiple layers of high purity, energy-intensive materials could increase environmental impact while reducing economic incentives from recycling. Producer take-back, when not mandated, will likely depend on ease of economics and, when regulated, may encourage greater design efforts to facilitate ease of recycling. The incorporation of thin-film PV into products with shorter lifetime may not reap EPBT benefit of recycling because they are less likely to be collected for recovery; this is challenge for future PV applications. Where unregulated, end-of-life PV will likely end up in municipal solid waste, where the likely regulatory body has more incentive to recover materials with mature recovery infrastructure and technology such as Fe, Al, and glass and less incentive to recover cell materials despite their high material value and energy-intensity. This is especially true for cells with increasing smaller material quantities but greater material diversity e.g. multi-junction PV.

4. TRADEOFFS IN SITING A SOLAR PHOTOVOLTAIC MATERIAL RECOVERY INFRASTRUCTURE

4.1 Introduction

Despite the steady increase in recycling rates over the last 50 years, 54% of municipal solid waste (MSW) generated in the U.S. ends up in landfills[141]. The majority (79%) of MSW generation is benign and relatively low economic value bulk materials for example, plastic, glass, paper containers, yard trimmings and food scraps. However, the rapid increase in the consumption of products that contain both valuable, e.g. indium, and hazardous e.g. arsenic, materials could increase the value, complexity, and toxicity of MSW. The increase in consumption of valuable and hazardous materials is a global phenomenon that has set off long-term material scarcity and waste management concerns [58,142] Recognizing the negative social, economic and environmental impacts of landfilling [143,144], policy is moving in the direction of waste reduction and waste elimination strategies that center around material recovery. This is especially the case for waste electronics and electrical equipment (WEEE) where state and national policymakers have recognized the potential of material recovery to address this compositionally complex waste stream. For example, at least 15 states, including California, have landfill bans on cathode ray tube (CRT) monitors and other WEEE, mainly because they contain lead. Although CRT monitors are an obsolete technology whose production peaked 15 years ago, its waste policy initiatives have only been enacted recently (earliest in 2009). WEEE landfill bans (which act as mandatory recycling policies) have been plagued by illegal waste exporting practices, negative environmental impacts of informal recycling, and difficulty integrating with the current MSW infrastructure [145-148].

The aforementioned challenges are not unique to WEEE. That is, all emerging technologies, especially those with potentially high volumes, that make use of valuable and hazardous finite material resources have uncertain material recovery routes. In this paper we use photovoltaics (PV) as an example to demonstrate a methodology for assessing siting tradeoffs of emerging technologies. Photovoltaics, despite rapid growth of nearly 100% per year, have not yet reached peak production. However, during the periods of its emergence and rapid adoption no end-of-life (EoL) policy has been developed in the US. Mature silicon and thin-film PV currently produced contain hazardous materials such as arsenic, zinc, and cadmium and valuable materials such as indium, gallium, and tellurium shown in Table 1. Due to the long lifetimes expected (i.e. greater than 20 years), lack of US EoL policy, and uncertain recycling technology, a looming waste stream of PV materials are predicted to confound the MSW systems [149].

Table 4.1 Primary Price and CERCLA Toxicity Score for Photovoltaic Materials

Material	Primary Price ($ per metric ton-module)	CERCLA Toxicity Score (out of 600)
Ag	5	53
Al	313	10
As	2	600
Cd	2	400
Cu	1	10
Fe	121	NA
Ga	73	10
Ge	213	53
In	256	53
Mo	9	53
Se	18	178
Si	128	NA
Sn	4	53
Te	16	53

Source: [56,62,150]

NA – Not applicable

Previous research has proposed various strategies and infrastructure configurations for EoL PV recovery that mimic other products, for example, extended producer responsibility for nickel-cadmium (NiCd) batteries, centralized second-party collection as for CRT monitors, and decentralized processing such as what exists for municipal solid waste. Although recycling has been proposed to reduce the lifecycle impact of photovoltaics, there is uncertainty about the environmental impacts of technologies employed and transportation required for recovery. Due to the spatial dispersion of PV and potential energy intensive thermal recovery process, recovery energy and emissions burdens may outweigh the primary energy savings of recycling. This tradeoff is particularly uncertain for lighter (frameless), low efficiency, low capacity modules i.e. thin-films. In addition, without mandates for collection and given low tipping fees, it is uncertain whether current MSW systems have economic incentive to recover PV at EoL. Another possibility is that since PV installations are apart of buildings, they may be categorized as construction and demolition (C&D) waste. Most states define C&D waste as "inert" and "uncontaminated" with a specific list of designated materials for example, Al, Fe, cement and glass. C&D material recovery facilities (MRFs) are typically equipped to recover only bulk metals i.e. Al and Fe. They would therefore be likely unable to recover the remainder of EoL PV materials i.e. Si, Te, Cd.

The approaches to the location-allocation problem for hazardous and MSW management attempt to answer: where do we locate facilities? And how do we decide to send, process, and allocate waste? Previous operations research literature has proposed multi-criteria decision analysis (MCDA) models that solve the location-allocation. These models include multi-objective decision models that optimize economic and environmental criteria[151-155]; whereas single objective models consider economic criteria alone[156,157]. Previous work has proposed novel approaches to integrate environmental criteria for example, Caruso *et al.* (1993)[153]uses scalar weights to optimize both economic, resource waste, and environmental impacts; Nema *et al.* (1999) links facility technology, risk of an accident, and residue generation in a hazardous waste facility setting; and Vaillancourt and Waaub (2002) includes implicitly spatial data e.g. proximity to residential areas to evaluate site environmental impact. Larger models showcase the ability to track internal and external material flows for example, Karagiannidis and Moussiopoulos (1998) [156] includes four levels of system hierarchy i.e. transfer stations, MRFs, landfills, thermal plants. Other work has integrated key stakeholders in MSW management such as Hokkanen and Salminen (1997) [152] developed a method for optimization irrespective of the number of decision makers and given imprecise data. Several works demonstrate the advantage of dynamic models for policy analysis for example Kirca and Erkip (1988)[158] interprets a dynamic problem as static using a multi-period model and Hu *et al.* (2002) [157] explores sensitivity to waste treatment requirements. These models lack the nuances of real-life MSW systems in that they do not link facility technology with material recovery rate nor explicitly integrate geographic information in decision-making beyond transport distances. In addition, these models lack an exploration of sensitivity of results to assumptions such as the collection rate or waste policy initiatives. These models also neglect to explore tradeoffs of centralized and decentralized MSW infrastructures.

Geographic information systems (GIS) literature has proposed several land suitability models to site a hazardous waste facility or landfill [159-164]. See [165] for a recent overview of this literature. Collectively, previous GIS models demonstrate the important tradeoffs of including qualitative and quantitative criteria for waste facility siting. However, they do not forecast the spatial dispersion of future waste generation nor analyze its impacts on site suitability. Filling this literature gap is important for increasing material recovery and shaping waste policy into the future.

Several questions arise from the previously mentioned literature gaps: how do we model material recovery for valuable and hazardous wastes with rapid growth and uncertain spatial dispersion? How do we evaluate the influence of spatial (e.g. land use) and non-spatial criteria (e.g. policy) on MSW system configuration? Model parameters such as cost, recovery rate, and land use will be explored to quantify the sensitivity of our model to economic, technical, and environmental assumptions.

4.1.1 The Study Area

New York State (Figure 4.1) is located in the northeastern and mid-Atlantic regions of US, bordered by Canada to the northwest, states New Jersey and Pennsylvania to the south and Massachusetts, Connecticut, and Vermont to the east. New York State contains 62 counties. Several bodies of water such as Lake Erie, Lake Ontario, Lake Champlain and the Atlantic Ocean border New York State. The tallest peak is Mt. Marcy at 1,623 km elevation located in the Adirondack Mountains. However, the remaining elevation is limited. New York State's land cover consists primarily of cultivated crops, pasture, woody wetlands, and deciduous forests. The state has a land area of approximately 141 thousand km2. At least 6,967 separate PV installations are located in the state [166]. The average size of installations size is 7.6 kW and the total state capacity (as of July 2014) is 135 MW. There are at least 130 MRFs currently within the state that process a variety of materials e.g. metals, glass, paper, plastic, and electronics. There are also 37 landfills currently located in the study area.

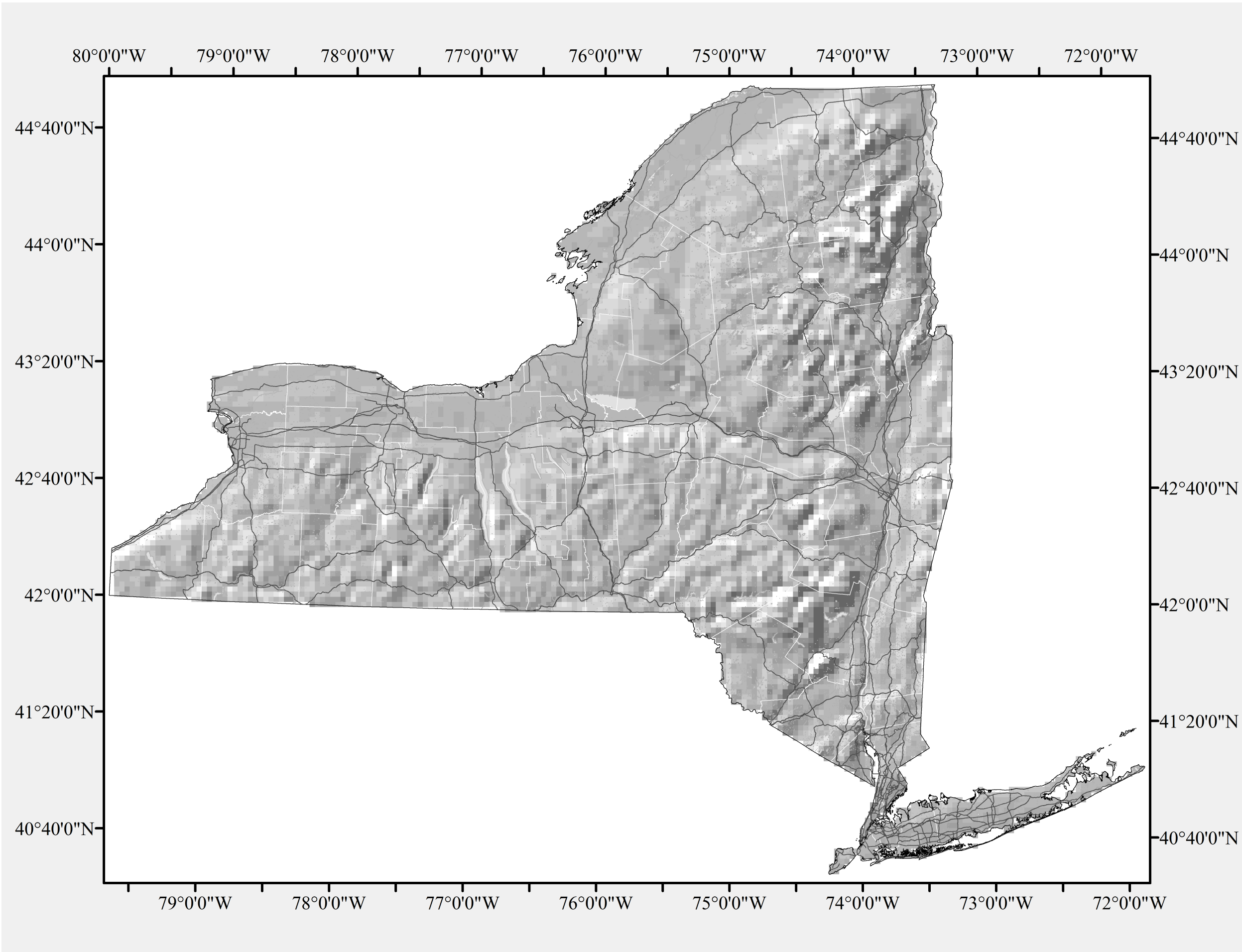

Figure 4.1 Study Area, Municipality Boundaries, Major Roads, Surface Water, and Elevation Features

4.2 Proposed Model Formulation

In order to address the questions posed above we integrated GIS and multi-criteria decision analysis (MCDA) tools to perform three key actions: spatial dispersion model (Section 4.2.1), land suitability (Section 4.2.2), and location allocation (Section 4.2.3) as shown in Figure 4.2.

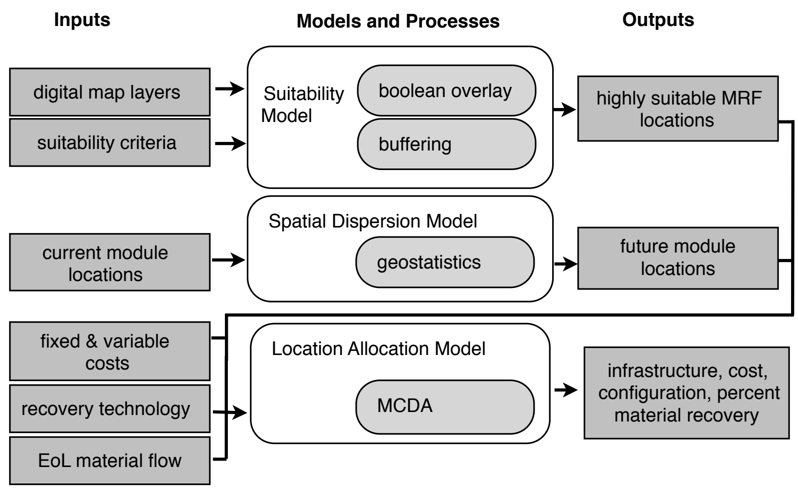

Figure 4.2 Model Relationships, Inputs, Outputs, and Embedded Processes

4.2.1 PV Spatial Diffusion

The purpose of our EoL PV diffusion model is to estimate the spatial dispersion and capacity of new PV modules. Previous GIS literature assigned PV dispersion based solely on technical, geographic, and environmental criteria such as land use, rooftop availability, solar insolation, population density and access to grid [167-174]. However, there are also empirical relationships between PV siting and policy or social criteria. For example, the share of PV capacity is largest among states that have both renewable portfolio standards and economic policy incentives such as subsidies for residents and businesses[175]. Recent work has also proposed PV installations are likely to be spatially clustered based on demographics or influences from peers and community organizations[176-178]. If we assume PV capacity is spatially clustered then we would expect new PV installations to be located close to existing installations. For simplicity, we assign new installations to the same location, e.g. XY coordinate or zip code area, as current installations. The size of new installations is set as equal to the average size of current installations at the same location. The total installation capacity at a point represents the neighborhood capacity rather than the likely capacity at the exact XY location. For computing efficiency in larger study areas, we define installation locations by aggregating points within a political boundary e.g. city or zip code. This approach to assigning new PV location and capacity attributes assumes that share of PV capacity is static over the same spatial area.

In order to test our assumption that PV capacity is spatially clustered we calculated the probability that a PV installation will appear in a location near an existing installation using spatial statistics. The use of spatial statistics for this purpose consists of four parts: Ripley's K-function, average nearest neighbor index (ANNI), p-value, and z-score calculations. Ripley's K-function has been used to determine spatial relationships in ecology[179,180] and biology fields[181]. The average nearest neighbor index has been used to determine clustering of epidemics, diseases [182,183], natural phenomenon[184,185], and population demographic densities [186,187]. Ripley's K function, like ANNI, is used to determine spatial clustering of n features that deviates from a random process. However, Ripley's K function summarizes spatial dependent over a range of distances, d for a weights $k_{i,j}$ (Equation 4.1). If there is no edge correction, the weight will be equal to 1 when the distance between i and j is less than d. Otherwise the weight will be zero. The function then provides insight in the range of distances for which statistically significant clustering, if any, occurs. ANNI compares the observed mean distance $\overline{D}_O$ between each feature and its nearest neighbor with the expected mean distance $\overline{D}_E$ for a feature in a random pattern (normal distribution) as shown in Equation 4.2[188]. The expected mean distance $\overline{D}_E$ is a function of n and the enclosing area A (Equation 4.3). In this case we use set the enclosing area as equal to that within state boundary. The z-score and p-value are two other statistics that determine whether spatial variables are random. The z-score utilizes the difference between the observed and expected mean distance to determine whether clustering is randomized (Equation 4.2). Random clusters would fall within two standard deviations from the mean of a normal distribution. The p-value determines the probability of obtaining the observed result assuming randomly dispersed features. P-value can be determined from z-score using z table for a 90% confidence interval. If any of the geo-statistical variables are statistically significant, then we reject the null hypothesis. The null hypothesis states that the spatial relationship between PV installations is random. We reject the null hypothesis for p-values less than 0.1, z values greater than 0.5, and ANNI less than 1.

$$L(d) = \sqrt{\frac{A \sum_{i=1}^{n} \sum_{j=1, j \neq i}^{n} k_{i,j}}{\pi n (n-1)}}$$

Equation 4.1

$$ANNI = \frac{\bar{D}_O}{\bar{D}_E} \quad \textbf{Equation 4.2}$$

$$Z = \frac{(\bar{D}_O - \bar{D}_E)}{0.26136}\sqrt{\frac{n^2}{A}} \quad \textbf{Equation 4.3}$$

$$\bar{D}_E = 0.5\sqrt{\frac{n}{A}} \quad \textbf{Equation 4.4}$$

This approach to PV spatial dispersion is limited in its capacity to forecast location specific adoption over time as well as to account for varying stages of adoption maturity. For example, if we assume the study area is in a mature phase of adoption, then the location of new installations can be predicted with greater certainty. However if spatial areas are in innovative early stage of adoption then, major, novel installations that may spring up where no PV currently exists may not be predicted by this model. It is also possible that clustering groups extend beyond regional or political boundaries are at different stages in product adoption.

4.2.2 Site selection suitability model for specialized MRFs

The purpose of this model is to develop optimal siting for specialized material recovery facilities (MRFs) that process end-of-life solar photovoltaic modules using the 15 map layers as shown in Table 1. Boolean, buffering, proximity and overlay methods in ArcGIS Desktop 10.1 were used to determine which sites were suitable for material recovery facility (MRF) siting. Available information related to the technical, environmental, and economic implications of MRF siting was obtained from several publicly available sources detailed in Table 4.2 and produced in digital map layers. Exclusionary and non-exclusionary criteria were applied to these map layers. Exclusionary criteria include elevation, land use, surface water, coastline, floodplain, fault lines, hospitals and schools. Exclusionary criteria are considered decisive factors that draw a boundary where MRF siting is unsuitable. We eliminate unsuitable sites by utilizing Boolean overlay method similar to previous literature [189]. Non-exclusionary criteria include proximity to surface water, coastline, major roads, waste production centers, and landfills. Non-exclusionary criteria rank suitability along a subjective, unit-less scale of 1(least favorable) to 10(highly favorable) based on proximity. Moderately suitable values were given a score of 5. Favorability rankings for each map layer were then linearly combined in an overlay method. In this study, each map layer was weighted equally. In practice, experts or stakeholders may weight map layers by priority. The areas deemed suitable were then further filtered out based on assumptions about the maximum socially acceptable facilities per municipality.

Land cover. There are 6 major classes of attributes specified in the land cover map layer: forest, barren land, cultivated crops, wetlands, pasture, and shrubs. Previous literature has suggested that landfills and recovery facilities avoid locating in industrial or agricultural areas [159,190,191] while others encourage the use of these areas [192]. Landfill leachate that contains arsenic, selenium, or cadmium has the potential to contaminate soil and crops as well as endanger wildlife flora and fauna. Therefore we strictly exclude siting in agricultural areas, wetlands, and forest while allowing siting on barren land, shrubs, and pasture land types. In addition, we increase favorability rating of site as proximity to these areas increases.

Surface water, coastline, floodplain, fault lines and elevation. Previous studies have used a buffer from rivers and streams of 0.1 - 0.5 km[161], and 0.5 km [189,193], 0.8 km [194], 1 km [159], and 2–3 km[195], We set distance of less than 1 km from surface water as exclusionary criteria. Unlike inland surface water buffers, coastline boundaries in New York State outline international drinking and fishing water sources of Lake Erie, Lake Ontario, the St. Lawrence River and the Atlantic Ocean. International law discourages “damage” of these waters [196]. Previous literature proposes 1 km [189,197] buffer from coastlines. We also set coastline buffer at 1 km while increasing favorability of siting as proximity to coastline and surface water decreases.

Floodplain and fault line information for this region is sparse. When available we follow [160,198-200] in setting 0.3 and 0.5 km distance from floodplains and fault lines as exclusionary, respectively. In regions where data is unavailable, previous work has set 3 km [190,199] as buffer from rivers and streams.

Previous studies elevation criteria for siting landfills widely vary, buffering at slopes less than 15 degrees [189,197], 30 degrees [160], 50 degrees [201], and between 8-12 degrees [202,203]. We set buffers at elevations between 0-12 degrees with 8-12 degrees being most favorable and 0 degrees being the lease favorable because the above research suggest landfill failures and contamination of water bodies from runoff can be controlled at these angles.

Waste production centers, landfills, current recovery facilities, roads, and site area. In literature, exclusionary buffer for residential areas is less than 1 km [192], 0.5 km[165,197,201], or 0.25 km[204]. However, the farther away MRF or landfill distance the greater the transport cost, therefore, literature has proposed upper bounds for the proximity to waste production centers of 60 km [197] or 5 km [165], or assigned a transport cost of 47 dollars per metric ton [198]. We set a proximity from waste production centers (current PV installations) of 2 km as most favorable and 50 km as least favorable, while excluding distances less than 0.5 km from residences and greater than 100 km from waste production centers.

For better integration into the existing infrastructure and to reduce transport costs, new MRFs should be located near current landfills, transfer stations, roads, and recovery facilities. Previous literature has argued that increasing proximity to roads minimizes cost and facilitates access. However, the flow of large vehicles carrying waste can also potentially hamper transportation in general and therefore exclusionary buffers of 0.2 km [161,189,194] and 0.1 km [201] have been proposed. Given previous literature, we set an exclusionary buffer of 0.2 km around roads. We also scaled favorability of siting to increase as proximity to roads, current landfills, and recovery facilities increase.

Following the identification of candidate sites, we determine the minimum site area to process the expected waste over the long term (50 years). Previous engineering literature suggests landfill sites require approximately 1 hectare of land for 80,000 metric tons of waste [205,206]. Likewise the site size for a MRF depends on waste capacity. Previous literature suggests MRFs requires between 11,380 and 1,080 square meters for 500 and 10 metric tons per day (TPD), respectively [207].

Schools, hospitals, airports, facilities per area, and vulnerable populations. Previous literature also suggest locating sites at a distance of 1.5 km from sensitive buildings such as hospitals [191] 200-800 meters from schools [208] and 3 km from airports[165]. Therefore, we set a buffer of 200 m, 3 km, and 1.5 km from schools, airports, and hospitals, respectively.

Siting should also be sensitive to oppressed and vulnerable populations. Environmental justice literature has recognized that low income populations are more likely to live in proximity to landfills and other sources of toxic releases partially due to political disempowerment [144]. Health literature has also documented higher rates of disease among the young (below age 2) and elderly (above age 65), especially for those in close proximity to sources of toxic releases [209,210]. In order to take into account the social impacts of siting, we use three population statistics where siting is restricted: greater than 25 percent elderly, greater than 25 percent of young, and a poverty rate above 50%.

Many social science studies have documented the negative attitudes of residents toward locating new landfills and recovery facilities nearby e.g. Not-In-My-Backyard (NIMBY) [193,211]. Since some municipalities may have several optimal sites for siting we limit the density of sites considered in any given municipality to less than one per 10 km^2 in order to reflect the likely NIMBY attitudes of residents.

Table 4.1 MRF and Landfill Site Selection Model Criteria Description by Category

Category	Map Layer	Criteria Description
Land Cover	Elevation	Slope less than 12 degrees
	Land cover	Areas with significant economic or ecological value should not be considered i.e. wetlands, forest, and cultivated crops
Hydrography	Surface water	Sites should be at least 1 km from rivers, streams, lakes, and ponds
	Coastline	Sites must be at least 1 km from coastline; increase favorability as proximity to coastline decreases; 1-5 km is least favorable. 6-10 km is moderately favorable; greater than 10km is highly favorable for facility siting
	Floodplain	Sites must be at least 3 km from streams
Geology	Fault lines	Sites must be at least 0.5 km from fault lines
Infrastructure	Roads Airports Hospitals Schools Landfills Installations	Sites must be at least 0.2 km from major roads and schools; 1.5km from hospitals; 3 km from airports; increase suitability as proximity from roads, landfills, and waste production centers increase; 0.2 - 10 km from is highly favorable; sites between 10 - 20 km is moderately favorable; more than 20 km is least favorable
Vulnerable Populations	US Census Blocks	Sites may not be located within census blocks that contain greater than 50% poverty rate or greater than 25%

		population that is aged over 65, or under 2 years
Site Density	Suitable sites	No more than one site per 10 km^2

Table 4.1 GIS input map data sources

GIS map layer	Scale	Map or Data Set Source
Elevation	1:24,000	NYS GIS Clearinghouse
Land Use	1:100,000	MRLC National landcover dataset
Roads, airports		ESRI
Population by age, poverty rate		US Census Tigerdata
Landfills, MRFs, transfer stations		NYS Department of Conservation
Schools, hospitals		NYS GIS Clearinghouse
Fautlines		US Geological Survey Quaternary fault and fold database

4.2.3 Location and Technology Allocation Model

Economic Model. In this section we propose a generic quantitative model for waste recovery network design. Our model is based on previous work on municipal solid waste recovery network properties discussed in the introduction that has demonstrated the utility of single-objective multi-period decision models that track internal and external material flows. Building on previous work, this model inputs spatial data from sections 4.2.1 – 4.2.2 and material data as described in Figure 4.2. The network has three levels - customer, facility, and landfill - and materials flow in a forward direction to either the materials market or towards landfill.

The structure presented in Figure 4.3 is translated into a mixed integer non-linear program (MINLP) in Equations 4-11. Here we modify the traditional facility location model by including technology decision variables and its recovery rate parameters γ_k for each facility. In this model there are I customers with end-of-life PV available for collection, J potential material recovery facility (MRF) locations, and K potential recovery technologies to use at MRFs. We are interested in X_{ij} the weight of EoL PV from customer i that is processed at MRF j with technology k. The variable cost parameters of this supply chain include c_k technology, c_t

transportation and c_d waste disposal while fixed cost of technology f_k and MRF facility f_j are also included. The facility capacity parameter is represented by z_j.

$$min \sum_{j \in J} f_j Y_j + \sum_{j \in J} \sum_{k \in K} f_k Y_{jk} + \sum_{j \in J} \sum_{k \in K} \left(\sum_{i \in I} X_{ij} \right) c_k Y_{jk} + \sum_{i \in I} \sum_{j \in J} c_t t_{ij} X_{ij} + \sum_{i \in I} c_d U_i$$

[Equation 4.5]

In this formulation the objective is to minimize the cost of the material recovery network. This objective (equation 4.5) is subject to constraints of mass balance (equation 4.6-4.7), facility & technology opening conditions (equations 4.8-4.10), capacity constraints (equation 4.11), and non-negativity (equation 4.12). This formulation is general and can reflect many different recovery scenarios. For example, we can model mandatory collection by setting the cost of disposal c_d to an extremely high or infinite value. We can also simulate a distributed or centralized system by setting $\sum_{j \in J} Y_j > 1$ or *j =1* respectively. In addition to capacity constraint, we can also simulate capacity size decisions. To do this we add the term $\sum_{j \in J} \sum_{z \epsilon Z} c_z Y_{jz}$ to the general objective function and modify the general capacity constraint (equation 4.11) as $\sum_{i \in I} X_{ij} Y_j \leq \sum_{z \epsilon Z} Y_{jz} z_j$. These new terms use a capacity cost parameter c_z and facility capacity decision variable Y_{jz}. Municipal solid waste networks also vary over time. In order to model network configuration dynamics we can either re-run the model each period with new waste material inputs or define all decision variables within time t. A network that is time varying would also require adding to the objective function a penalty $p(Y_{j,t} - Y_{j,t-1})$ for closing a facility that was open in the previous period.

$$\sum_{j \in J} X_{ij} + U_i = d_i \qquad \forall i \in I \; [\, Eq. 4.6]$$

$$U_i = d_i - \sum_{j \in J} X_{ij} \, Y_{jk} + \sum_{j \in J} \sum_{k \in K} X_{ij} \, Y_{jk} (1 - \gamma_k) \qquad \forall i \in I \; [Eq. 4.7]$$

$$\sum_{i \in I} X_{ij} \leq Y_j \qquad \forall j \in J \; [Eq. 4.8]$$

$$Y_{jk} \leq Y_j \qquad \forall k \in K \; [Eq\ 4.9]$$

$$Y_{jk}, Y_j \in \{0,1\} \qquad \forall j \in J, \forall k \in K \; [Eq\ 4.10]$$

$$\sum_{i \in I} X_{ij} Y_j \leq z_j \qquad \forall j \in J \; [Eq \; 4.11]$$

$$X_{ij}, Y_{jk}, Y_j \quad \forall i \in I, \forall j \in J, \forall k \in K \; [Eq \; 4.12]$$

The generic model has the following assumptions:

- The network configuration is static i.e. one period model.
- Every specialized MRF can process all materials in a waste stream.
- Landfills will accept all waste materials at a penalty that linearly increases with material weight.
- Every facility must choose among a predetermined set of recovery technologies with a recovery rate of γ_k. The material recovery rate is dependent only on the recovery technology of the facility it is sent to.
- Once materials are put into the waste stream they are immediately processed, therefore, there is no material inventory at a facility, the only storage is at the landfill.
- Once materials arrive at landfill they cannot be recovered; likewise once materials are recovered and sent to secondary materials markets they cannot be disposed of.
- There are no stops along the route from a waste collection point to landfill or MRF. Waste transporters will travel only the shortest route between these stops.
- MRFs, landfills, and secondary markets are co-located, so no additional transport is necessary from MRF to landfill or MRF to secondary market.
- There is no cost to transport waste to landfill from a collection point. The disposal cost linearly increases with the mass of unrecovered waste. The transport cost from waste collection point to MRF varies linearly with distance. This implies that the size and the number of modules does not directly influence transport cost.

For the base case we assume:

- Transportation cost due to diesel fuel of 0.99 $/L for a 7.5 metric ton capacity truck with 3.85 km/L fuel economy.
- Each facility has 10TPD capacity that requires of $0.51 M fixed construction cost.

- The proportion of capacity at a site will remain unchanged. For example, a site containing 5% of current capacity, will remain as such in the future. Assuming the study area will develop a total PV capacity of 6,000 MW, a site containing 5% current capacity will develop 300 MW capacity over the long term.
- Available material at each location is proportional to the capacity such that 85 W capacity module contains approximately 8 kg of materials.
- There is no cost to transport materials to landfill however there is a disposal cost at the landfill of $60 per metric ton.
- The study area will develop a PV capacity of 6,000 MW.

Despite its flexibility there are several aspects of material recovery infrastructure not taken into account with this model. In particular, the temporal uncertainty of EoL modules, which we previously stated is a potentially important aspect of reducing the energy intensity of future photovoltaics, is not modeled explicitly. We address this issue for our case study by analyzing the impact of supply uncertainty on technology and facility location using sensitivity analysis.

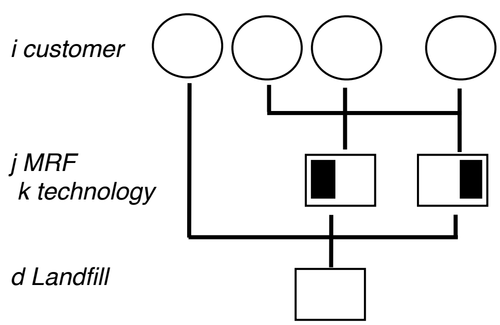

Figure 4.3 Material Recovery Infrastructure Hierarchy and Material Flows

Recovery Technology. The degree of recycling and recovery technology used is uncertain despite sparse efforts of module manufacturers such as First Solar and recent European Union (EU) electronics laws mandating recycling for PV modules [83]. There are a variety of processes being developed and currently employed by module manufacturers that vary in cost, methods, and material recovery priority. For example, [212] describe technology meant to recover silicon modules for remanufacturing. Due to the high manufacturing energy requirements of silicon modules and lack of active secondary market for silicon, this technology has the potential to reap the most primary energy benefits including lowering module energy payback time. Other technologies are aimed at recovering rare expensive module materials such as tellurium despite the high processing costs and uncertain economic benefits [78]. There is also the option of exclusively recovering module glass and frame materials as they reduce the energy payback time by the greatest amount due to their high module compositions [150]. However, depending on available technology and costs, this option may potentially result in landfilling the small amounts of critical and hazardous metals.

In the location allocation model, it is assumed that all MRFs have the option of two recovery paths: (1) limited recycling and (2) exhaustive recycling as shown schematically in Figure 4.4. The "limited" recycling path recovers frame, glass, and laminate materials by manual disassembly, thermal processing, size reduction, leaching, and sieving steps. The "exhaustive" recycling path includes etching, precipitation, and thermal processing in addition to the processes included in the limited route. The technologies employed for each path are module dependent as shown in Table 6. In each recycling path, recovered materials will be sent to scrap markets whereas waste materials will be sent to landfill. In the case of limited recycling, active materials and unrecovered glass are sent to landfill. in the case of exhaustive recycling, dust and unrecovered glass is sent to landfill.

Each module type (i.e. silicon-based or thin-films) also undergoes separate processing steps summarized in Figure 4.4 and Table 4.2. Silicon-based modules first are manually dissembled of frame materials, then undergo thermal and chemical processing before being crushed in order to recover frame, silicon, and glass materials following the Deutsche Solar process described in literature [213-215]. These processes have yielded up to 90% and 95% recovery of silicon and glass, respectively. Thin-films CIGS and CdTe modules first will be shredded, spun, and then undergo leaching and precipitation in order to separate out ethyl vinyl acetate (EVA) and organics while recovering glass and active materials following the First Solar process described in literature [78,80,81,216,217]. This process has reported yields between 90 – 95% Cd, Te and 99% glass from CdTe modules. A similar process that additionally incorporates electrolysis after sieving yields 94% Cu and 88-90% Se from CIGS modules[99]. The above literature suggests the same results can be obtained by replacing the energy intensive electro-winning step with multiple precipitation cycles. There are few literature sources exploring a-Si recycling methods and costs, therefore, we assume this module type will be recycled using the same steps as other silicon-based modules.

Each recovery process and path has variable costs, fixed equipment costs, and process efficiency assumptions based on data from literature (Table 4.3). Processing cost for thin-films (T1 – T4 in Table 4.3) based on Choi and Fthenakis (2010) include utilities, waste treatment, overhead, maintenance, tools, and consumables. Processing cost for silicon-based modules (S1 – S2), based on a bench scale process described by Frisson et al. (2000), include utilities, consumables, waste treatment, and labor. We assume a large commercial recovery facility will be able to improve costs over bench scale by 70% based on energy technology learning research [218]. To obtain process costs per metric ton, we assume each module has 72 wafers. In addition to the process specific equipment, we assume a forklift and conveyor belts are required for transporting materials throughout the plant. Conveyor belts are assumed to be used to feed into size reduction and spinning processes as described in literature[217]. We estimate the equipment cost is \$1,500 and \$9,000 for a conveyor system and forklift based on data from equipment manufacturers[219,220]. The variable cost is \$2.5 per metric ton for the forklift based on Choi and Fthenakis (2010). For processing step S0, we assume PV module frames can be disassembled in 3 minutes for a labor wage of \$9 per hour. Overhead costs are 1.5 times total labor cost.

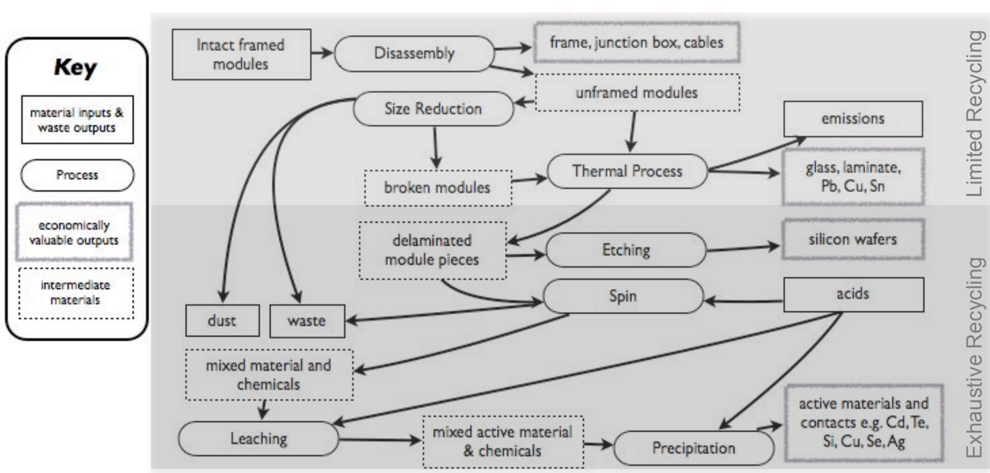

Figure 4.4 Limited and Exhaustive Recycling Path for MRFs

Table 4.2 Thin-Film Module Recovery Description, Fixed Equipment and Process Costs

ID	Process Name	Description	Processing ($/ton)	Equipment Name	Equipment ($100)
T1	Size Reduction	separate glass from laminate	22.2	Shredder Hammermill	100 100
T2	Spin	semiconductor film removal from glass substrate using slowly rotating drum with acids	58.9	Rotating drum Classifier	20 10
T3	Sieving	solid/liquid separation, glass rinse	18.2	Screen 100 L Tank	50 5
T4	Precipitation	precipitation of precious metals	22.5	100 L Tank	5
S0	Disassembly	detach frame, junction box, and cables with hand tools	36	Hand tools	3
S1	Thermal	glass/laminate separation, neutralize furnace emissions	138	Furnace After burner	50 300
S2	Etching	remove Ag material from wafers	209	100 L Tank	5

4.3 Results

4.3.1 Spatial Dispersion and Land Suitability

The average nearest neighbor index, z-value, and p-value are 0.17, -132, and zero, respectively. The observed and estimate mean distance are 387 m and 2554 m. These results indicate that PV installations in the study area are not randomly (normally) dispersed. Ripley's K function, agrees, determining a statistically significant clustering for point distances between 0.3 and 0.5 km. This result validates our decision to assign new wastes streams of EoL of current PV installations in anticipation that future installations will be spatially clustered with current installations. This method is valuable to waste planning organizations looking to include waste production centers as a part of land suitability especially when waste dispersion is uncertain.

After applying both exclusionary and non-exclusionary land suitability criteria we have determined that there are at least seven sites that are highly suitable for MRF siting (labeled F1-F7 in Figure 4.5). These sites are all either co-located (within 1 km) of a landfill or current MRF. The co-location of current and planned future waste facilities serves as a validation of the siting model. Of the planned sites, F2 has the least combined road distance to all current PV installations. F7 is the closest to large capacity installations.

The land suitability analysis determined that the majority of the study area is either unsuitable or poorly suitable for the siting of an MRF as shown in Table 4.4. Non-exclusionary criteria reduced the suitable areas by 86%. This is largely due to environmental criteria: land cover and hydrography. For example, surface water and cultivated crops accounted for 54% and 33%, respectively, of study area unsuitability. The high density of roads in the densely populated urban areas also resulted in 15% of study area unsuitability. There is a great deal of overlap between exclusionary features due to the size of the buffer criteria and close spatial relationships e.g. roads and schools. The observed criteria redundancy also suggests that reducing exclusionary criteria to fewer, more meaningful categories may achieve similar results. For our study area, reducing exclusionary criteria from 11 to 3 features i.e. roads, cultivated crops, and surface water achieves 90% of the original suitability area.

The land suitability method implies a higher priority for social and environmental considerations because 80% of the criteria pertain to either environmental or social features. Changing the priority of economic, social, and environmental criteria impacts our land suitability results. For example, increasing the weight of economic criteria in the weighted overlay analysis increases the availability of sites near urban centers, landfills, and roads. Decreasing the number of environmental criteria such as hydrology increases the availability suitability sites near floodplains, rivers, and surface water. The weighting of criteria has broad implications for the perceived suitability of sites. Stakeholder and expert input should be gathered before priority and criteria decisions are applied to siting.

Table 4.3 Suitability Ranking by Total and Percent of Study Area

Suitability Ranking	Area (km2)	Percent Area
Highly Favorable	5,031	4%
Moderately Favorable	11,320	9%
Poorly Favorable	1,258	1%
Unsuitable	107,766	86%

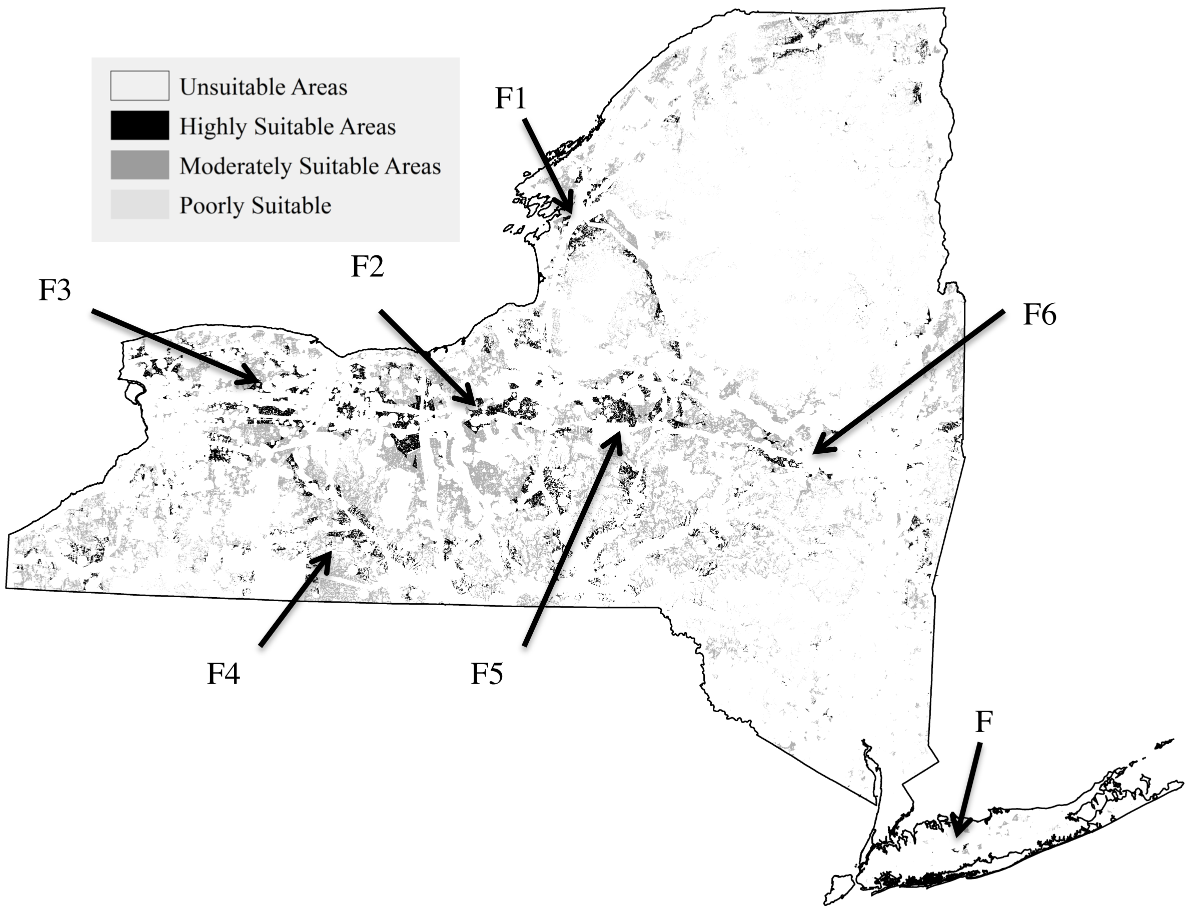

Figure 4.5 Map of Suitability Rankings: Unsuitable (white), Poorly Favorable (light grey), Moderately Suitable (dark grey), and Highly Suitable (black)

4.3.2 Location Allocation and Waste Policy Analysis

The allocation of materials for recovery depends on the fixed and variable costs of disposal, technology, construction, and transportation. For the base case, the lowest cost solution ($0.5 M) is to allocate all materials to landfill. Decreasing the construction or technology costs, individually, does not change this result. It is only when either the disposal cost is increased or the transport cost is decreased that the model decides to allocate materials to newly sited MRFs. Decreased transport cost implies lower fuel cost and greater vehicle fuel economy which may require smaller trucks and more trips. An increased disposal cost implies increased landfill tip fee, which is set by municipal waste managers. As we increase the tip fee, the collection rate and total system cost increases as shown in Figure 4.6. This result indicates that tip fee may be utilized as a policy mechanism to incentivize material recovery. However, this requires coordination to avoid the negative environmental impacts of greater waste transport. For example, waste haulers conscious of high tip fees in one municipality, may decide to transport waste further to a cheaper landfill, thereby increasing transport emissions. From our case study, statewide disposal costs would need to increase to $1350 per metric ton in order to achieve 50% collection rate for recycling. This would require a substantial cost increase of current landfill tip fees in the study area, which range $50-120 per ton for C&D, WEEE, and MSW.

There are multiple tradeoffs between variable costs, fixed costs, and recovery rate. The recovery rate is directly influenced by technology and material allocation model decisions. For the base case, with tip fees above $1300, the model chooses a limited recycling technology path. A limited recycling path recovers 95-99% by mass of a discarded module. This path leaves behind critical, and valuable materials such as Ga, Si, In, and Te. The exhaustive recovery path would recover the remaining material whose primary value of $20-350 per ton-module is PV technology dependent.

Constraining the model to meet a minimum collection rate increases material allocation to MRFs as shown in Figure 4.7. As the collection rate increases from 0 to 40%, total, fixed, and disposal costs per metric ton decrease. The lowest system cost is achieved at 40% collection rate. After 40%, the total, fixed, and transport costs per ton increase. As more materials are allocated to MRFs, more MRFs open, which increases the fixed cost of the system. Likewise, the first materials allocated are those closest (less than 50 km) to the active MRFs, which are the small (less than 5kW) residential installations. For the study area, the large (greater than 500 kW) commercial installations are clustered between 50-250 km from the nearest MRF, therefore, as the collection rate increases above 40%, the transport costs increase. In reality, large concentrated commercial installations are more likely to be the target of waste policy rather than dispersed residential installations. Therefore, siting and waste policy should take into account the likely source of materials given collection rate targets.

Despite the use of a minimum collection rate, the model decides to follow the limited recycling technology path. The model chooses an exhaustive recycling technology path only when its cost is equal to the limited recycling technology minus the difference in disposal cost. This case exposes the problem of the lack of coordination between waste policies. The use of a single-action waste policy e.g. tip fees is insufficient to drive exhaustive material recovery. In order to achieve exhaustive recovery, waste policies should be coordinated with optimal tip fees, collection rate targets, and subsides for recovery technology.

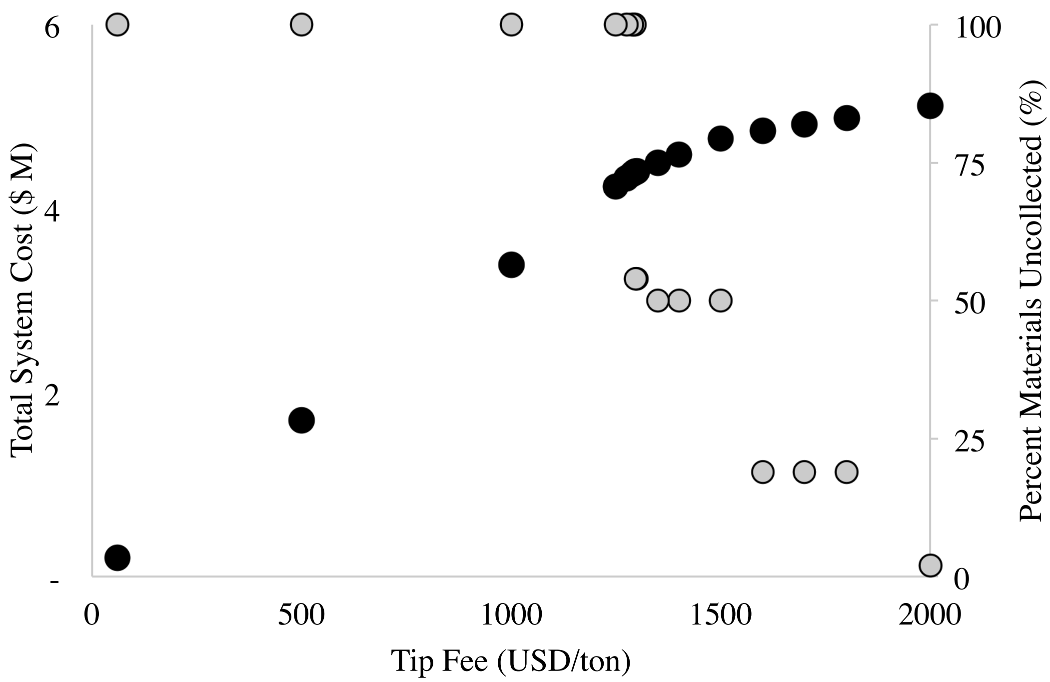

Figure 4.6 Total Cost (grey dots) and Percent Uncollected Materials (black dots) as a Function of Tip Fee

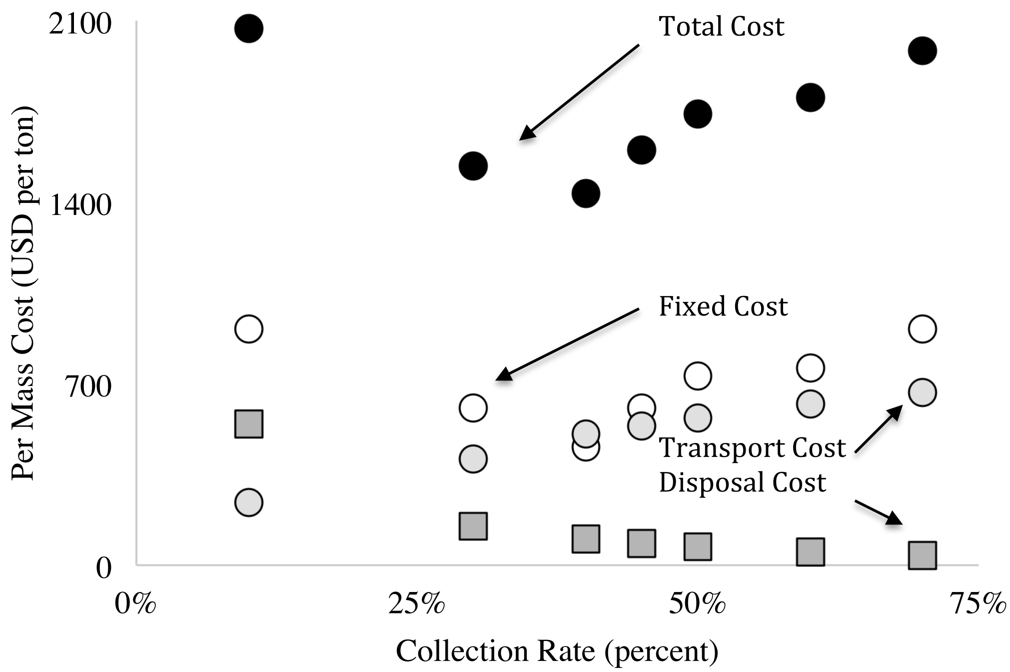

Figure 4.7 The change in per unit total cost (black dots), transport (grey dots), disposal (grey squares), and fixed cost (white dots) for a given collection rate

4.4 Conclusions

In this paper we have explored the social, economic, and environmental tradeoffs of siting a material recovery infrastructure for photovoltaics. We developed an approach that combines multi-criteria decision analysis with GIS tools. For the case study, the results indicate that PV installations are spatial clustered. At least seven sites, which are co-located with landfills and current MRFs, were 'highly' suitable for siting according to our criteria. After applying exclusionary criteria to the study area, 86% was deem unsuitable for siting while less than 5% is characterized as highly suitable. This method implicitly prioritized social and environmental concerns and therefore, these concerns accounted for the majority of siting decisions. As we increased the priority of economic criteria, the likelihood of siting near ecologically sensitive areas such as coastline or socially vulnerable areas such as urban centers increased. Accounting the direct environmental impacts of technology and siting decisions has not been done and therefore is currently under investigation by the authors.

The results of the location allocation model suggest that coordinated policy action is required to encourage the recycling of photovoltaic materials since the cost of disposal is lower than the cost of material recovery. In particular, our model estimates a tip fee $2000 per metric ton would achieve nearly 100% collection rate. However, the results show that exhaustive recovery requires a multi-pronged approach that lowers technology costs, imposes a minimum collection rate, and implements higher tip fees. Future work in this area may also include evaluating social, economic, and environmental tradeoffs of exporting photovoltaic waste for recovery overseas.

5. CONCLUSIONS

This dissertation emphasizes the important economic and environmental tradeoffs of recycling from three perspectives: supply risks (Chapter 2), cumulative energy demand (Chapter 3), and recovery infrastructure (Chapter 4). Specifically, in Chapter 2, we demonstrated active materials have a higher supply risk imperative for recycling but a lower per module recycling priority as compared to frame and conductive materials. This is especially the case for thin-film PVs, which have high bulk material and primary energy yet low per module value and lifecycle energy. The difference in recycling priority on a per module basis occurs because of low compositions among thin-film active materials. In contrast, Si, the active material of c-Si PV, has high per module priority but low bulk material priority for recycling. This is due to high per module composition of Si in c-Si PV.

Our investigation in Chapter 2 also revealed, for the PV materials studied, recycling was correlated with higher geopolitical stability, decreased foreign reliance, and primary price. That is, the higher the material recycling rate the lower the net import reliance, and the lower the potential for supply disruption due to violence or political instability. Recycling rates are also higher for expensive materials such as gold and platinum. However, material toxicity, in the US, is not a strong motivator for recycling. Ultimately, this work developed a tool for policymakers to utilize multi-metric analysis in policy analysis for critical materials.

In Chapter 3 we investigated recycling solar PV from a cumulative energy perspective. We found that there is an energy benefit to recover solar PV frame and mounting materials i.e. iron (Fe) and aluminum (Al). Especially for low efficiency (thin-film) modules whose potential reductions in energy payback time outpaced high efficiency modules. However, many thin-film modules do not contain frames; instead, they are encapsulated between two panes of glasses. This means frameless thin-film modules may not receive the EPBT gains of framed c-Si modules that are recycled. Overall, this work suggests recycling can achieve more rapid gains in energy payback time than efficiency improvements due the relative time required for each.

In Chapter 4, we investigated recycling from a recovery infrastructure perspective. We found that, our siting methodology produced available facility locations near current landfills and material recovery facilities (MRFs). In addition, due to the absence of waste policy that deals specifically with solar PV, the lowest cost solution is to landfill all PV materials. Only after the tip fee is increased does the model decide to recover frame and mounting PV materials only.

Active materials impose a greater economic burden on the recovery infrastructure due to high technology costs. Therefore, a combination of minimum collection rate, increased tip fee, and technology cost reduction policies are required to encourage exhaustive PV recovery. If recovery activities can be allocated by weight, then the majority of landfill costs and transportation energy for thin-films is due to glass. We speculate, due to relatively high recycling rate of glass and large number of facilities that recover glass in our study area, recycled PV glass may be more easily integrated into current municipal solid waste recovery infrastructure than other non-active materials.

Deepening the work of Chapter 3, explore the limits of secondary energy scaling assumptions in the energy payback calculation. Future work related to Chapter 4 could investigate the social and environmental implications of recovery infrastructure decisions such as transport, recovery technology, and landfilling. In addition, future work could utilize dynamic models to estimate material availability, PV adoption, and PV spatial dispersion. Lastly, work in the context of critical materials from a developing economy's perspective could extend the work of Chapter 2.

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